CHAPTER 1

Background

WHAT ARE SEAGRASSES?

Seagrasses are unique marine flowering plants of which there are approximately 60 species worldwide (den Hartog 1970, Phillips and Menez 1988). With the exception of some species that occur in the rocky intertidal zone, they grow in shallow, subtidal or intertidal unconsolidated sediments. Thus, they bind millions of acres of shallow water sediments in the coastal waters with their roots and rhizomes while simultaneously baffling waves and currents with their leafy canopy (Ginsberg and Lowenstam 1958, Taylor and Lewis 1970, den Hartog 1971, Fonseca et al. 1983, Fonseca 1996a). In this manner the canopy inhibits resuspension of fine particles and traps water-column-borne material (Ward et al. 1984, Short and Short 1984), clearing the water column. This cleansing effect extends to water column nutrients as well. Nutrient uptake by seagrass blades and their associated epiphytes and macroalgae as well as roots incorporate dissolved nutrients into plant biomass, which can improve water quality (Harlin and Thorne-Miller 1981). The baffling effect of the canopy on sediment stabilization is enhanced by the presence of a robust root and rhizome mat, although the relative contribution of the mat has not been isolated from canopy baffling in its role of sediment stabilization (Fonseca 1996a). The physical stability, reduced mixing and shelter provided by the complex seagrass structure provides the basis for a highly productive ecosystem (Wood et al. 1969). Overall the importance of seagrasses and their role in many coastal ecosystems has been extensively documented (see reviews by Thayer et al. 1975, Phillips 1982, Zieman 1982a, Thayer et al. 1984, Zieman and Zieman 1989) and the nature of their general function and high resource value are no longer an issue.

Seagrasses occur in all coastal states of the U.S. with the apparent exception of Georgia and South Carolina where freshwater inflow, high turbidity and tidal amplitude combine to prevent their occurrence. There are at a minimum thirteen species of seagrass currently recognized to occur in U.S. waters (Table 1.1). The presence of a fourteenth species, *Zostera asiatica* on the West Coast remains a subject of debate (Phillips and Wyllie-Echeverria 1990). We will not include in this discussion seagrass species occurring in U.S. possessions in the Pacific Ocean because little is known about their status; through NMFS Southwest Regional Office reports, we know that *Enhalus acoroides* and *Halodule uninervis* occur on Rota Island and Saipan Island in the Pacific Territories. Also, Phillips and Menez (1988) list *Halophila ovalis* and *Halophila minor* (fifteenth and sixteenth species) as species that occur in Hawaii. *Halophila hawaiiana* is also reportedly present on Hawaii (K. Bridges, Univ. Hawaii, pers. com.). Drawings of the major U.S. species are given in Figure 1.1. One species, *Halophila johnsonni* was only recently described as a separate species despite its occurrence in the heavily-studied region of southeast Florida. Because of its limited distribution, this species is currently under consideration for listing as a threatened species as defined by the Endangered Species Act. Another species, *Zostera japonica* was recently introduced to the Pacific Northwest . It is spreading and tends to colonize shallow intertidal flats, converting them from their historical ecological status as mudflats to intertidal eelgrass habitat (Harrison and Bigley 1982, Pawlak 1994).

Although recognized for their value where they occur, the distribution of seagrass is not as well known as it should be for proper management (Wyllie-Echeverria et al. 1994a). Moreover, knowledge of population-level temporal dynamics is only rudimentary at best. We know that at least 90 percent of the southeast United States seagrass acreage (~1.1 million hectares) exists in the Gulf of Mexico (Orth and Van Montfrans, 1990). But nationally, the distribution and abundance of two genera in particular have been overlooked. The full extent and function of the reported ~400,000 hectares of seasonal *Halophila* beds off the west coast of Florida (Iverson and Bittaker 1986) is unknown. Similarly the distribution of the Hawaiian *Halophila* is not reported. Also, very little is known about local distribution (distribution meaning localized, specific locations of beds, not the range of a species) of a unique West Coast dominant, the rocky intertidal *Phyllospadix* spp., although work has been done regarding its population ecology (Turner 1985, Turner and Lucas 1985). The distribution of seagrass on the West Coast, including both Alaska and Hawaii, has not been systematically compiled to the degree seagrasses have on the east and Gulf coasts

Table 1.1. List of seagrass by family, genus and species, and common names (if given) that are found in the United States and adjacent waters. Species marked with (?) are not fully documented as occurring in U.S. waters.

Family, Genus, and Species	Common Name[a]
Hydrocharitaceae	
Enhalus acoroides Royle	
Halophila decipiens Ostenfeld	paddle grass
Halophila engelmanni Ascherson	star grass
Halophila hawaiiana Doty and Stone	*Hawaiian* seagrass[a]
Halophila johnsonii Eiseman	Johnson's seagrass
Halophila minor (Zollinger) den Hartog?	unknown
Halophila ovalis (R. Brown) Hooker f.?	unknown
Thalassia testudinum Konig	turtlegrass
Potamogetonaceae	
Halodule wrightii Ascherson	shoalgrass
Halodule uninervis?	
Phyllospadix scouleri Hook	*Scouler's* seagrass
Phyllospadix torreyi S. Watson	*Torrey's* seagrass
Phyllospadix serrulatus Ruprecht et Ascherson	surfgrass
Ruppia maritima L.	widgeon grass
Syringodium filiforme Kutz	manatee grass
Zostera japonica Ascherson et Graebner	*Japanese* eelgrass
Zostera marina L.	eelgrass
Zostera asiatica?	*Asian* eelgrass

[a] Italics on common names indicate suggested common names; R. Phillips, Battelle Laboratories, Richland, Wa., pers. com.

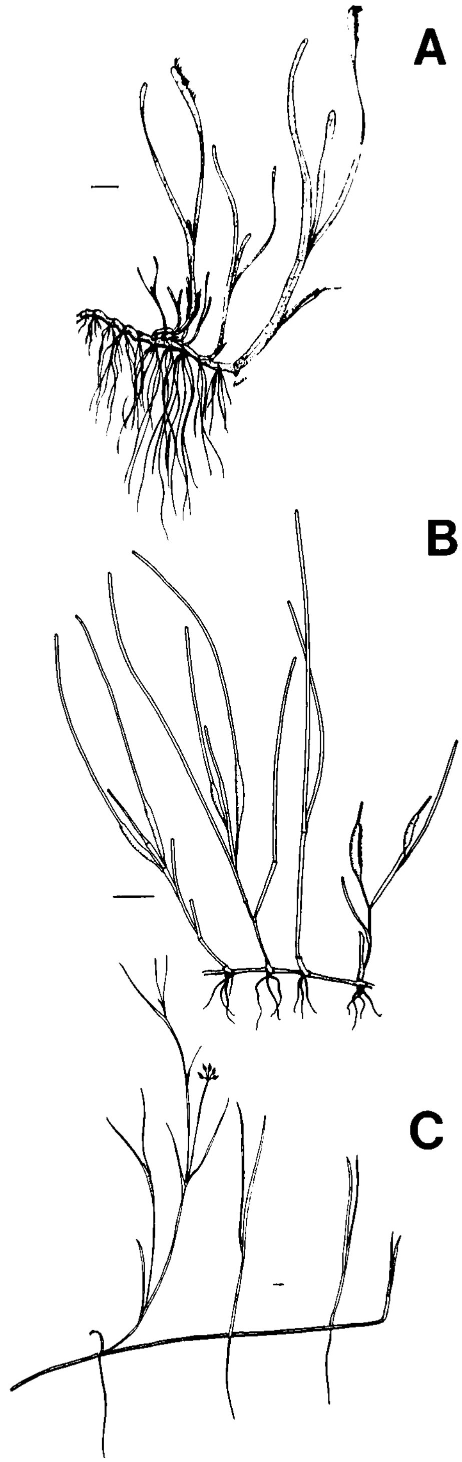
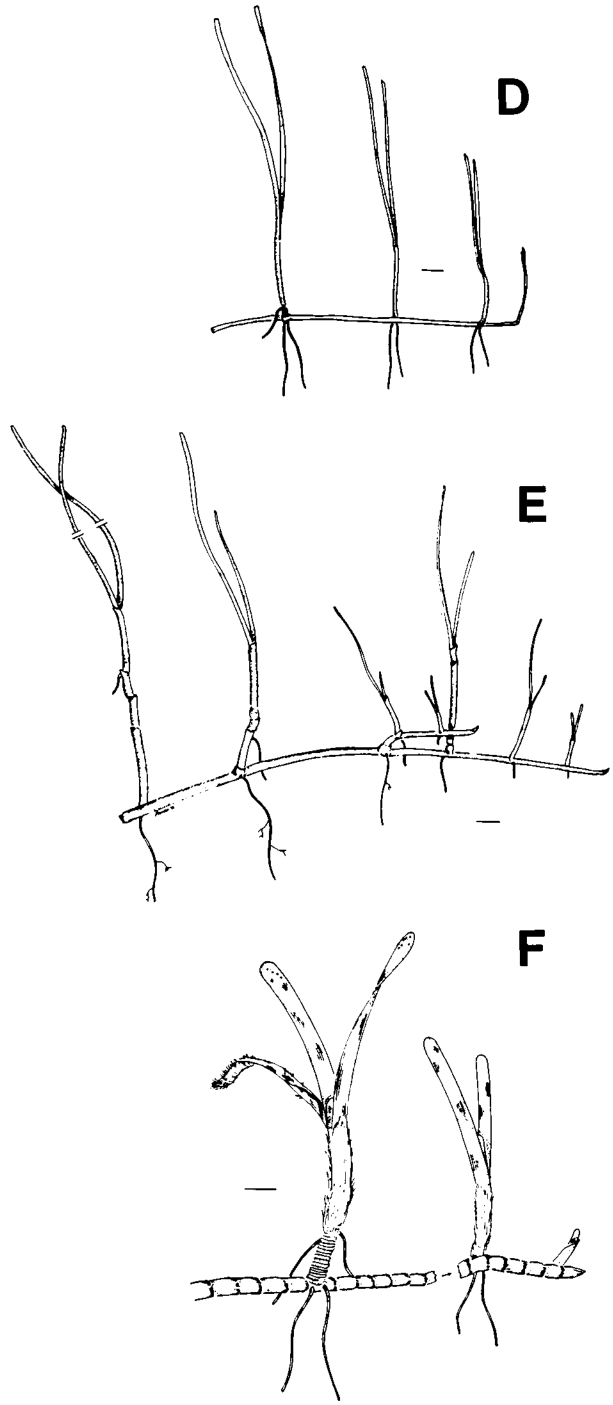


Figure 1.1. Drawings of most seagrasses found in U.S. waters (taken from Phillips and Menez 1988 and Fonseca 1994). All scale bars are set at 2cm and thus vary with seagrass species. A=Zostera marina*; B*=Zostera japonica*; C*=Ruppia maritima*; D*=Halodule wrightii*; E*=Syringodium filiforme*; F*=Thalassia testudinum*; G*=Halophila engelmanni*; H*=Halophila decipiens*; I*=Halophila johnsonni*; J*=Phyllospadix serrulatus*; K*=Phyllospadix torreyi*; L*=Phyllospadix scouleri.



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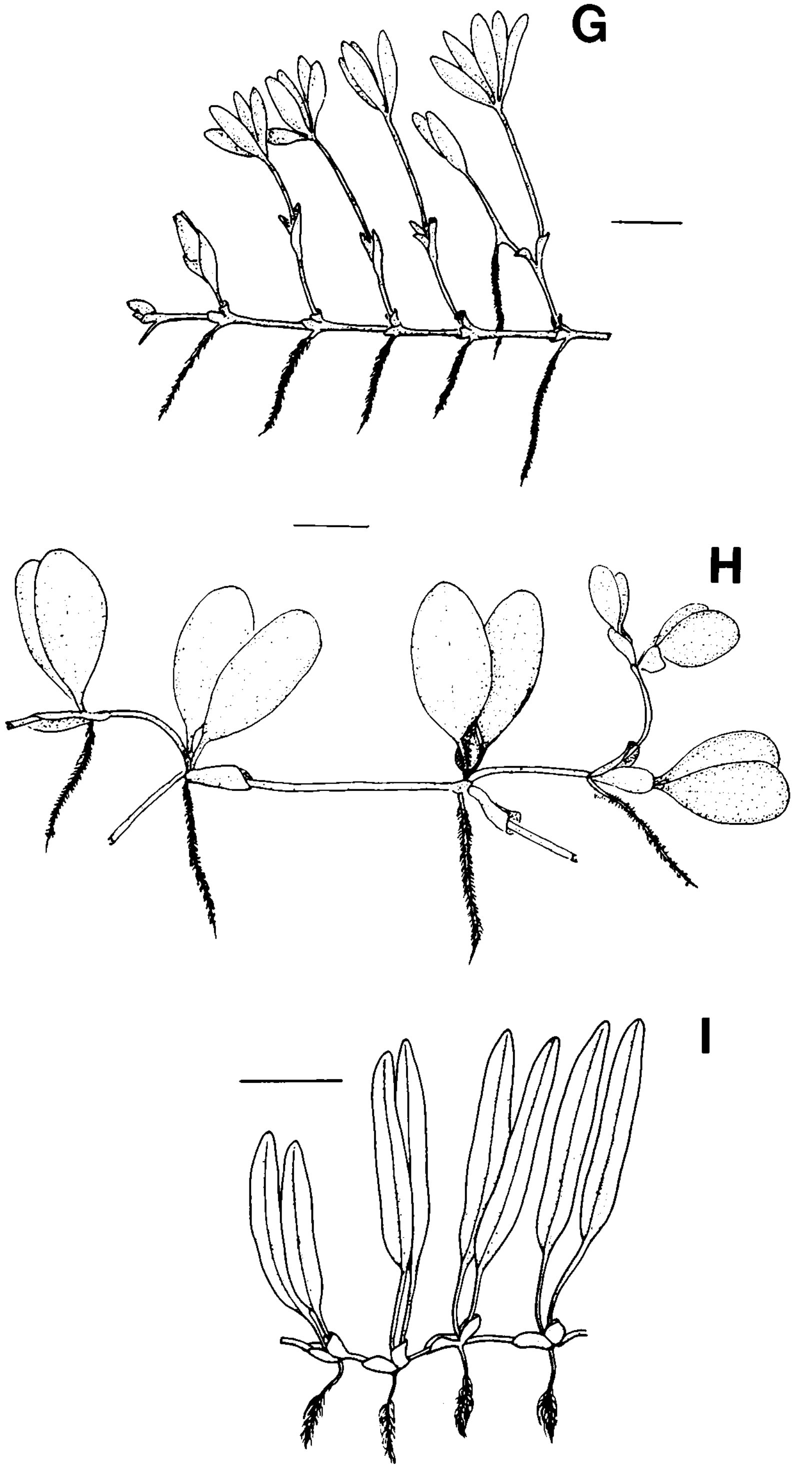

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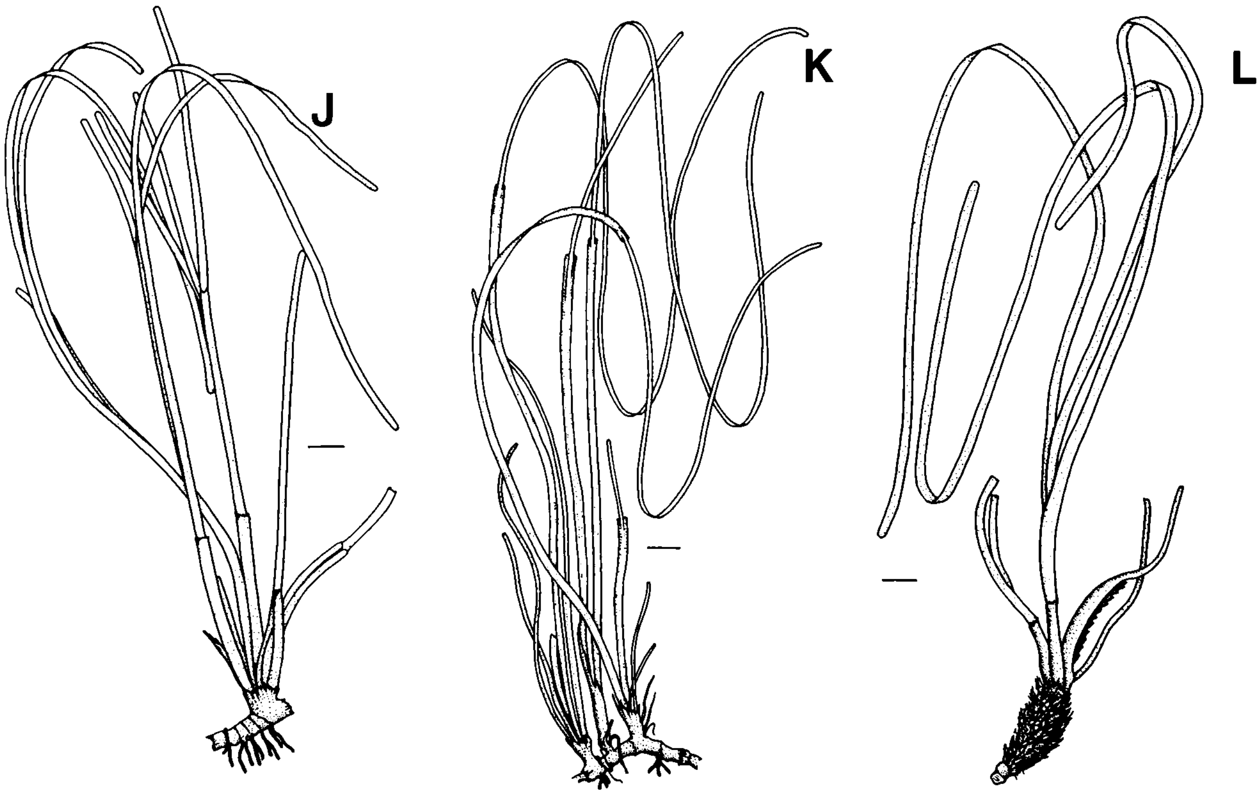

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although the general range of species' distributions has been reported (Wyllie-Echeverria and Phillips 1994).

Historically, emphasis been placed on aspects of seagrass primary and, to a lesser degree, secondary production attributes (see descriptions in Zieman 1982a, Phillips 1984, Thayer et al. 1984). Extensive information is available regarding light and nutrient requirements of seagrasses (Kenworthy and Haunert 1991, Dennison et al. 1993, respectively). Seagrasses are flowering plants and much attention has been paid to the mechanics of pollination and seed dispersal (see review by Cox 1993 and references therein) but much less is known about the role of seeding in bed maintenance or colonization of new areas (Kenworthy et al. 1980, Harrison 1993, Orth et al. 1994). With the exception of some recent studies (Duarte et al. 1994, Durako 1994) and previous transplanting data sets (Fonseca et al. 1987c), demographic studies have been sorely neglected in this country yet this is a topic area where managers ask many questions: How quickly will a seagrass bed recover from a given impact? Is planting necessary? Given intrinsic recovery rates and transplanting success, how do we compute replacement ratios or estimate interim loss? Should we be concerned about genetic diversity of the population? These questions are only now being addressed.

DEFINING SEAGRASS HABITAT

Seagrass beds exist in a wide variety of physical settings that lead to different coverage patterns. The problem is coming up with a consistent definition of what constitutes a seagrass bed. Although small patches may themselves have significant resource value, how does one assess the collection of patches and determine the boundaries of a seagrass habitat? Seagrasses exhibit a variety of growth strategies and coverage patterns which occur from rocky and soft-bottom intertidal habitats to depths of at least 40 meters. Some species can rely heavily on seeding to ensure year-to-year survival (e.g., *H. decipiens* and possibly *H. engelmanni*) meaning that surveys during winter months would need to include sediment seed bank assessments to accurately define the presence of a seagrass bed. Moreover, some species, such as *Z. marina,* can exist either as perennials or annuals, again requiring very different assessment strategies, varying between seed bank and vegetative material depending upon time of year. Clear knowledge of seagrass population ecology is a requirement for effective management and planting; that is, one-time snapshot inventories are a very, very poor basis upon which to delineate seagrass habitat.

Seagrass beds move. Depending on the species and the physical setting, the rate at which portions of the seafloor switch from vegetated to unvegetated may vary on the scale of days or decades, meaning that the amount of open seafloor required to maintain patchy seagrass beds is greater than the coverage by the seagrass itself at any one point in time (Figure 1.2), sometimes by a factor of two (i.e., over time, the movement of seagrass beds means that they will soon occupy at least twice the presently unvegetated bottom evident at any one survey time). Thus, if unvegetated areas among existing patches of seagrass are converted to channels, the long-term (within four years, unpubl. data) baseline acreage of seagrass in the vicinity of the converted habitat, will decline. Therefore, seagrass habitat must be recognized as including not only continuous cover beds, but chronically patchy habitat; a policy that requires considering the (presently) unvegetated spaces between seagrass patches as seagrass habitat as well. Management of seagrass resources therefore depends on understanding the spatial and temporal dynamics of seagrass coverage.

One of the biggest problems regarding delineation of seagrass habitat relates to the choice of sampling scale during the process of inventory, especially prior to a planned impact to a seagrass bed (see section, "Spatial Scale and its Role in Defining Seagrass Habitat," below). Scale is roughly defined here as the variation of pattern as a function of the range and resolution of examination. The scale at which assessments of seagrass coverage take place varies tremendously, depending on some covariate of acreage, interest and time available to conduct surveys. In contrast, after a planting is installed, monitoring of seagrass plantings is less prone to scale problems as direct count methods are usually employed and statistical sub-sampling protocols can be instituted to ensure adequate sampling intensity. However, assessment of existing natural seagrass and post-coalescent seagrass plantings takes place at many spatial scales and this leads to very different values of seagrass abundance. If aerial photographs are used, the altitude of the airplane, the camera lens, film, solar angle, water turbidity, and wind waves affect the ability to detect seagrass beds, particularly at the lower end of their depth distribution. Similarly, if one chooses to survey a potential impact site from the deck of a small boat then wavelets, reflectance, turbidity and an individual's search image all influence ability to assess seagrass abundance. Aerial photography such as that recommended by the NOAA Coastal Change Analysis Program (C-CAP) (Dobson et al. 1995), has a minimum mapping unit of 0.03 ha. At that resolution roughly 37 percent of the permits issued for alter-

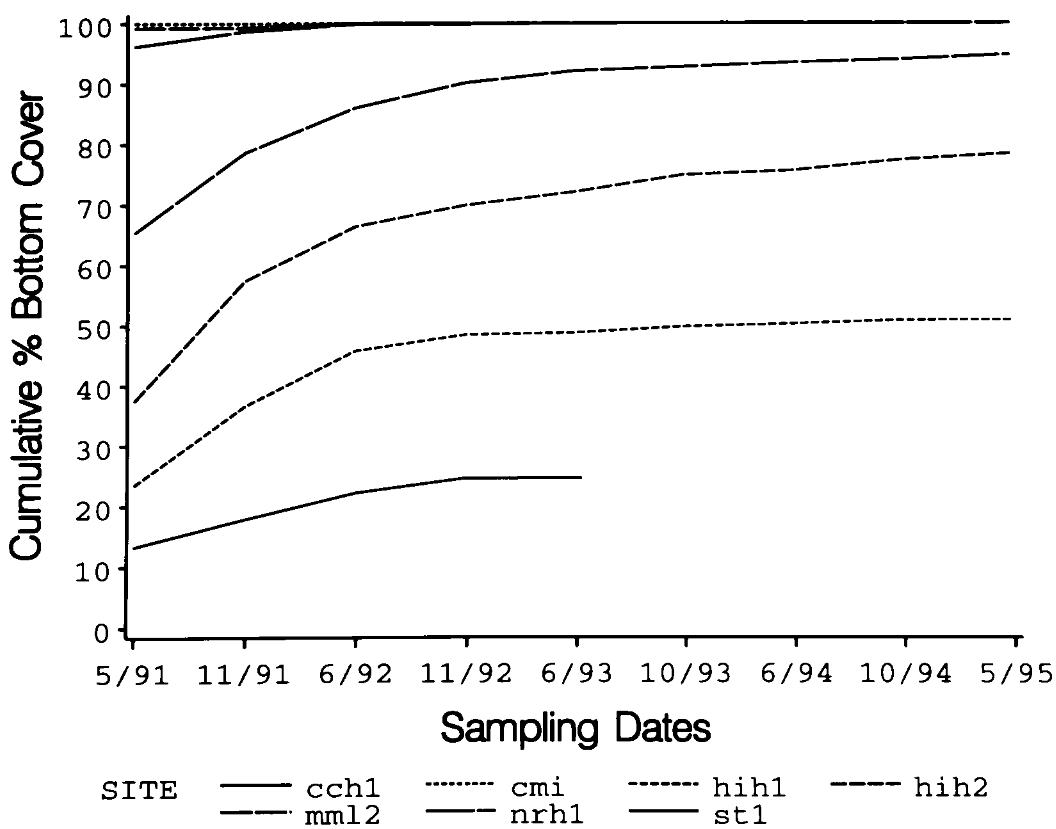

Figure 1.2. Plot of the cumulative area of bottom covered in 50 x 50m survey areas over time. Y-axis = cumulative cover assessed by adding new square meters of cover to that not previously covered in any survey. X-axis = sampling dates. Each line type represents a different 50 x 50m site.

ation of submerged aquatic vegetation habitats could not be detected. Fortunately, those that could be detected with 0.03 ha resolution accounted for ~99 percent of the acreage impacted (Rivera et al. 1992). Inherently patchy seagrass beds would be even more difficult to detect and quantify at a spatial resolution less than 0.03 ha using C-CAP techniques. These scales < 0.03 ha are spatial scales that questions of planting unit (PU) spacing and groupings of PU must be addressed (see section on "Spacing of Planting Units"), and persistent seagrass patches can be produced at these smaller scales.

Fonseca (1989a) suggested that at the 1:24,000 scale of aerial photography when the ratio of average seagrass patch diameter to the distance between patches exceeds 50:1, seagrass habitat continuity no longer fosters cognitive recognition by a viewer as constituting seagrass habitat. He suggested that above that ratio the area should no longer be considered continuous seagrass habitat. Clearly this ratio is scale dependent. If a ratio of 50 shoot widths to the distance between shoots were used, then many seagrass beds on the West Coast and in the northeast where individual plants are very large (> 2 m length) would no longer be considered seagrass habitat even though the unit area biomass might be comparable to other seagrass beds in the country. Unfortunately, we are not aware of any quantitative description of how bed boundaries are interpreted (i.e., when a bed is drawn as one large polygon or many small polygons). However, variation in seagrass bed form can easily be visually detected from low-level aerial reconnaissance (Figure 1.3), and appears to be correlated with exposure to waves and currents. Under wave and current conditions beds can take extreme forms; Molinier and Picard (1952) and Fonseca (1996a) described vertical walls of *Posidonia* and *Zostera*, respectively, revealing the extent to which seagrass could reduce erosion and enhance sediment accumulation. Seagrass patterns also change, revealing areas of seagrass coverage loss and gain at meter scales within short time periods (months) (Figure 1.4) attesting to the consistent ability of seagrasses to stabilize sediments. For at least 20-30 years after Molinier and Picard's work, little in the way of a quantitative association of seagrasses' effects on water motion and, conversely, the effect of water motion on seagrass bed development took place. During this time, interest in the physical processes occurring in seagrass beds was confined largely to qualitative descriptions of their geological role and, to a lesser degree, the implications of this geological stability on animal utilization.

It is unlikely that there will be a universal standard for defining seagrass habitat. Different seagrass species form beds that occupy too great a diversity of habitats and exhibit such a range of life history strategies that a universal definition would almost certainly be restrictive and unworkable. Further, published data on seagrass biomass, density, and structural complexity (e.g., surface area) have tended to be collected from

Figure 1.3. Aerial photograph of mixed Halodule wrightii, Ruppia maritima, Syringodium filiforme, *and* Thalassia testudinum *beds on the western margin of Tampa Bay, Florida. In the foreground at the bayward edge of the shoal are what appear to be wave-sculpted beds while further landward, in shallow water are more continuous cover bed. Reduction in wave energy from both the shelving shoal and the grass itself is thought to be responsible for the resultant seagrass bed landscape pattern. Taken from Fonseca (in press).*

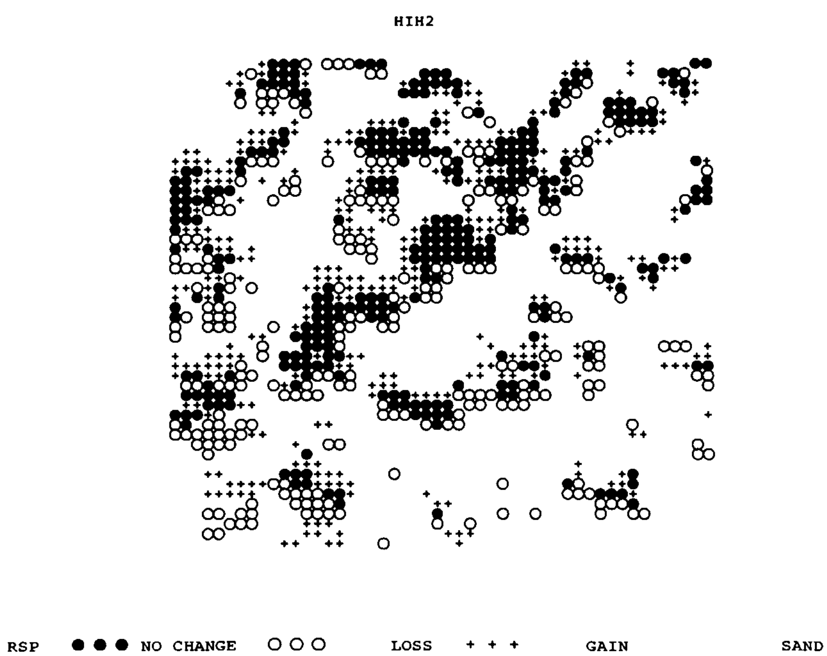

Figure 1.4. Change in seagrass bed cover in a wave-exposed, patchy seagrass bed near Beaufort, NC. Dark circle=m^2 areas with no change in cover (6-month period), + = areas of seagrass gain, ◦ = areas of loss and no symbols = areas of unchanged sand.

seagrass beds that form large unbroken meadows. Limited comparative information on bed spatial heterogeneity is available from the full range of habitats or landscape patterns that seagrasses form. Therefore, if we used published data to set boundary definitions of seagrass beds, it is quite likely they would tend to define only certain seagrass species (i.e., commonly studied species such as *Z. marina*, etc.) in certain settings (e.g., relatively wave-protected and low current speeds which yield extensive, non-patchy habitat). Further, because data collection has been historically biased toward beds in lower energy environments, the more fragmented, patchy nature of higher energy seagrass beds would be an element of seagrass bed structure that would not be captured in such a universal definition. On-site, direct surveys of local undisturbed seagrass beds in similar physical settings, or better, pre-impact surveys of the status of a seagrass bed remain over time the best guidelines for delineating seagrass habitats.

What we suggest is that managers must have some historical perspective. One-time surveys are completely inadequate data (i.e., see Figure 1.2) upon which to base management decisions that could have effects for years. Bed form migration (*sensu* Patriquin 1975, Marba et al. 1994, Marba and Duarte 1995), presence of seed banks, annual populations, recent nonpoint source anthropogenic impacts (e.g., decreased water clarity), and even deliberate removal of seagrasses all combine to cast doubt on the veracity of one-time surveys (i.e., see Figure 1.2). For evaluations of extant beds, even seemingly straightforward information such as shoot density can be misleading. Data such as shoot density are sometimes inversely related to shoot size, meaning that shoot densities of even less than one shoot m^{-2} may be significant, especially if that shoot is very large. Conversely, populations of *Halophila* spp., of which there may be in excess of half a million hectares in the Gulf of Mexico and Indian River Lagoon (Iverson and Bittaker 1986, Continental Shelf Assoc. 1991, Kenworthy 1992), return almost exclusively from seed every spring (Williams pers. com.). As with other species that rely heavily upon seeds for seasonal recovery, surveys taken during months where aboveground biomass is all but absent and that do not incorporate seed bank surveys would erroneously conclude the area did not support seagrass.

SPATIAL SCALE AND ITS ROLE IN DEFINING SEAGRASS HABITAT

If physical processes have the potential to affect habitat heterogeneity in seagrass communities then there is the potential for affecting associated fauna (Fonseca and Fisher 1986). Seagrass beds composed of isolated, dune-like patches of ~2 m in diameter can coalesce within several growing seasons upon elimination of waves and tidal currents (pers obs). Despite the clear relationship of water motion to seagrass bed form, we have only begun to evaluate their spatial (or temporal) organization (Virnstein 1995), otherwise seagrass beds have consistently been treated as a "black box" at the landscape scale. To build on information accumulated on ecosystems and apply this information to seagrass systems, research emphasis must include not only the normative 1 m scale study, but scales that are relevant to mechanisms that contribute to the formation, maintenance, and function of whole systems, such as sediment transport pathways or an organism's range.

If the pattern of distribution observed in seagrass beds is the result of physical processes whose effects vary with the spatial scale of examination, then it follows that the influence of bed pattern on such things as faunal abundance will, in turn, vary with spatial scale as well (*sensu* Bian and Walsh 1993, Fonseca 1996). Therefore, knowing the range of these scales is potentially valuable if, after gathering empirical evidence, one can infer structural attributes at other scales of interest, especially scales that may be less expensive to derive (e.g., aerial photography).

Resource managers must realize that a relationship between ecological phenomena and the spatial scale of a survey is real and sometimes intuitive. At the least, such relationships are a statistical reality that can strongly affect interpretation of field survey data (Rossi et al. 1992, Cao and Lam 1997). The notion that interactions at one scale (spatial or temporal) affect that which is expressed on another scale provides the basis for hypothesizing scale-dependent effects. Therefore, spatial and temporal patterns seen in seagrass ecosystems are the result of physical processes acting both on individual plants and the local population level (individual patch). Responses of individual plants to water motion and associated phenomena (e.g., sediment particle size) may be cumulative and affect seagrass landscape patterns perceived at coarser scales of resolution. To summarize, examples of the importance of deriving scale dependence in seagrass beds include identification of:

1. The scale at which samples taken in the landscape are independent of one another and improve sampling stratification,

2. Their effect on animal utilization and distribution, and

3. The relevant scales over which sedimentary processes are controlled providing a better prediction of alterations in current patterns, interception (or lack thereof) of wave energy, and sedimentary processes as the result of altering the seagrass landscape.

One result of recent research on seagrass landscape patterns is that there are ranges of spatial scales over which estimates of coverage vary as the result of the scale of sampling resolution chosen by the investigator (Fonseca 1996b). Moreover, for seagrass beds in North Carolina and Tampa Bay that experience relative wave exposure values (see "Constraints Imposed by Physical Setting on Planting Operations," below) greater than 3×10^6 (on a scale that runs from 0 to ~ 6×10^6) any estimate of seagrass coverage will differ depending on the size of the sampling unit and/or the distance separating those sample units at scales < 10m (Fonseca 1996b). This means that interpretation of any factors related to seagrass bed coverage sampled within this range of 1-10 m will be different among any studies that sampled at different spatial scales (i.e., samples taken 1 m apart versus, for example, 5 m apart). Therefore, comparisons among studies or surveys, even of the same bed, will differ to some degree simply because different size quadrats were used and not necessarily as the result of actual differences in the factor being compared. Of course, comparisons between studies can be different because different numbers of samples (which approximates statistical power) are taken. Finally, this has implications for the integrity of sampling schemes because any samples taken in this range of scale dependence will not be statistically independent, casting doubt on the validity of among-study or among-survey comparisons which were conducted at different spatial scales. This can create problems for interpretation of planting success.

Scale dependence in sampling has not only spatial but temporal considerations. We raise this caution regarding temporal scale dependence because in our section titled "Comparative Analysis of Seagrass Planting Efforts" we found that many projects changed assessment frequency during the course of the monitoring period. In fact, we too recommend a change in assessment protocol depending on whether it is being conducted before or after coalescence of planting units. Therefore, statistical comparisons should be made with caution between data collected from pre- and post-coalescence because such comparisons of one site over time likely violate rules of sample independence. Because many planting projects cannot escape problems with sample independence over time, the use of simple descriptive measures (such as area covered and persistence) as standard measurement protocols becomes very important to minimize problems with comparative analysis among studies or among dates within studies.

Another problem with spatially heterogenous (i.e., patchy as opposed to continuous) seagrass beds is the perception of their comparative ecological function. Spatially heterogenous seagrass environments in North Carolina have been classified as "scattered" (Carraway and Priddy 1983) versus continuous cover beds that are termed "dense." This unfortunate classification inferred a lower resource value despite the fact that the former landscape pattern covers many thousands of acres of estuarine seafloor in North Carolina, has shoot densities and primary production equivalent to continuous cover beds, has significantly higher below-ground biomass than continuous beds, and often supports equal densities of some economically valuable species such as pink shrimp (Murphey and Fonseca 1995).

Vulnerability and Susceptibility of Seagrass Ecosystems

Why are seagrasses so often impacted by human activity? One of the reasons is their location in the coastal zone. Because of their relatively high (compared to phytoplankton) light requirements (Kenworthy and Haunert 1991) they occur in shallow, nearshore waters, a situation that makes them extremely susceptible to damage by human activity such as nutrient loading (Short and Burdick 1996), light reduction (Dennsion et al. 1993, Kenworthy and Fonseca 1996), and propeller scarring (Sargent et al. 1995). As our utilization of the coastal zone grows so will the damage to seagrass ecosystems unless proactive steps are taken to avoid those impacts and successfully mitigate when impacts occur. Because they are now universally recognized to be valuable habitats, efforts to mitigate their losses have been underway for many years.

It is critical that one recognizes that seagrass mortality, whether mechanically induced, such as dredging, or physiologically induced from reduction in light (e.g., docks, turbidity), often happens rapidly; time scales for loss can range to as little as weeks or months. Recruitment, however, does not typically keep pace, yet if the site were capable of supporting continued cover, seagrass may recolonize within a few growing seasons (Kenworthy et al., 1980, Harrison 1987, Fonseca et al. 1990, Thayer et al. 1994). Recovery via natural recruitment is a demographic process with tremendous spatial and temporal variation (e.g., 0 to > 10,000 seeds m^{-1} for *Z. marina*) and is very difficult to predict. It is clear, however, that seed set and successful germination are often requisite for *rapidly* (1-2 growing seasons) balancing anthropogenically induced seagrass mortality. In contrast, vegetative encroachment may take many years (Johannson and Lewis 1992) or even longer, as is suggested by the lack of seagrass recovery in portions of the northeast U.S. from the "wasting disease" loss of the 1930's (*sensu* Short et al. 1993). The point here is that there are fundamentally different time scales involved in population-scale losses and their recovery. Only recently have investigations begun to assess the population-scale processes of seagrass bed formation and maintenance (Orth et al. 1994). In fact, scientists have no clear idea what constitutes a population for these plants or what population processes are at work (i.e., existence of metapopulations, *sensu* Orth et al. 1994). At a minimum, documentation of distribution together with elucidation of demographic process must be a research priority.

HISTORICAL IMPACTS AND LOSSES

We have mentioned environmental constraints to seagrass planting (see review by Phillips 1982), but there are many other management constraints that determine the effectiveness of seagrass planting. One is the degree of philosophical alignment among federal, state and local agencies whose jurisdictions include seagrass habitat. The U.S. Army Corps of Engineers, whose function includes issuance of dredge and fill permits, sometimes cannot follow recommendations from other agencies to conserve seagrass habitat (Mager and Thayer 1986). Conflicts between preservation of seagrass (and many other wetland habitats) and implementation of public-interest development projects must be balanced by resource agencies but often results in the loss of seagrass habitat (*sensu* Race and Fonseca 1996). The loss of seagrass habitat is sometimes addressed by proposing in-kind mitigation. In addition, maintenance dredging projects, particularly those associated with national security, are often considered exempt from mitigation requirements although in instances of very long dredging cycles (years to decades), mitigative actions are sometimes implemented to minimize immediate impacts. It has been our experience that as more information is presented to managers regarding the functions of seagrass ecosystems and the difficulties involved in mitigating for their loss, fewer permitted impacts are occurring in seagrass beds.

Although the loss of seagrasses due to dredging has been significant (Taylor and Saloman 1968, Onuf 1994), it is likely that the majority of seagrass habitat loss does not result directly from dredge-and-fill activities. More recently, direct impacts from mooring scars (F. Short, Jackson Est. Lab., Durham. NH, pers. com.), propeller scars (Sargent et al. 1995), jet skis (Kreuer pers. com.) and vessel wakes (pers. obs.) are emerging as a major source of seagrass habitat loss. For some species of seagrass such as *Thalassia* which is slow spreading (Fonseca et al. 1987c), physical damage is extremely long-lasting (Zieman 1976, Durako et al. 1992). Short et al. (1993) and the Chesapeake Bay Program (1995) recognized improvement of wastewater treatment, surface run-off, restrictions on certain fish and shellfish harvesting techniques, and regulation of boat traffic as key elements in protecting seagrass beds. Although scallop harvesting has been shown to damage seagrass beds (Fonseca et al. 1984) as has raking (Peterson et al. 1984) and prop-dredging for clams (Peterson et al. 1987), other fishery techniques such as trawling for bait-shrimp with specially-designed gear can have little apparent effect on seagrass although by-catch mortality is severe (Meyer et al. in review). Work by the Chesapeake Bay Program (1995) also lists (blue) crab dredging (scraping) as a significant impact on eelgrass beds. Fishing gear impacts to seagrass beds must be examined on a gear-by-gear basis.

Reduction in water quality, including water clarity, is another significant agent of seagrass loss (Dennison et al. 1993, Gallegos 1994, Onuf 1994, Gallegos and Kenworthy in press). Burkholder et al. 1992 and Dennison et al. (1993), like Batiuk et al. (1992), provided general guidance on maintaining water chemistry to support healthy seagrass beds. In doing so, Dennison et al. (1993) essentially determined the converse of health standards; they defined some critical water chemistry conditions at which harm would come to seagrass beds (Table 1.2). These data, and those promulgated by the Chesapeake Executive Council (1989) and the Chesapeake Bay Program (1995), are perhaps the only quantitative water chemistry information for managers to evaluate the health of seagrass environments at this time. They are likely useful for most temperate seagrass ecosystems and likely describe levels that would be too high for typically oligotrophic tropical and sub-tropical waters, particularly those dominated by carbonate sediments (*sensu* Fourqurean et al. 1995). However, the correlation between human development of the shoreline and seagrass decline is clear (Short and Burdick 1996).

Although seagrass beds are dynamic systems, with some beds persisting essentially unchanged for decades, others change with the season (den Hartog 1971, Zieman and Wood 1975, Phillips 1980a, Fonseca et al. 1983, Duarte and Sand-Jensen 1990). Some changes in seagrass communities can be attributed to the life histories of individual seagrass species (e.g., *Halophila* spp.). However, natural perturbations

Table 1.2. Chesapeake Bay submersed aquatic vegetation habitat requirements. For each parameter, the maximal growing season median value that correlated with plant survival is given for each salinity regime. Growing season defined as April-October, except for polyhaline (March-November). Salinity regimes are defined as tidal fresh = 0-0.5 o/oo, Oligohaline = 0.5-5 o/oo, Mesohaline = 5-18 o/oo, Polyhaline = more than 18 o/oo. (Taken from Dennison et al. 1993).

Salinity regime	**Light attenuation coefficient ($K_d m^{-1}$)**	**Total suspended solids (mg/l)**	**Chlorophyll *a*(*u*g/l)**	**Dissolved inorganic nitrogen (*u*M)**	**Dissolved inorganic phosphorus (*u*M)**
Tidal freshwater	2.0	15	15	-	0.67
Oligohaline	2.0	15	15	-	0.67
Mesohaline	1.5	15	15	10	0.33
Polyhaline	1.5	15	15	10	0.67

greatly influence the distribution of seagrass species. Disease has been widely implicated in the loss of seagrass beds since the pan-Atlantic decline in the 1930's (Rasmussen 1973, Short et al. 1987, Muehlstein 1989). Through this time, seagrass declines attributed to disease have added significantly to fluctuations in seagrass distribution. Physical disruption from storms and shifting channels redefine seagrass bed distribution and composition. Seasonal disturbances, such as low tides which expose and desiccate beds (Phillips 1980a, Thayer et al. 1984), and catastrophic events, such as hurricanes (Eleuterius and Miller 1976, Livingston 1987), can dramatically restructure seagrass beds both in terms of bed size and seagrass species composition. We have found that reductions in seagrass bed coverage as the result of storms is a positive function of how exposed to wind-generated waves a bed is prior to a storm; rapid loss of coverage can occur within a period of hours (unpubl. data), reiterating the fact that one-time surveys of seagrass coverage can be misleading as to the potential distribution of seagrass in a water body.

Biological disturbance of seagrass beds by a variety of organisms can also be extensive. Overgrazing by herbivores such as urchins has also affected spatial distribution and standing stock of seagrass beds (Camp et al. 1973). Ice scour (Robertson and Mann 1984) and extreme cold (Lalumiere et al. 1994) have been shown to control *Z. marina* distribution in the sub-Arctic. Also, excessive epiphytic load (Sand-Jensen 1977), burrowing shrimp (Suchanek 1983), vagile macrofauna (Valentine and Heck 1990, Valentine et al. 1994), green algae (den Hartog 1994a), and lugworms (Philippart 1994) have all been shown to limit seagrass distribution (but see Reusch et al. 1994; fertilizer enhancement of eelgrass by blue mussel biodeposition). Rays too have been implicated in many seagrass planting failures (Merkel 1988a, Mote Marine Laboratory and Mangrove Systems Inc. 1989, Fonseca et al. 1994) and may even contribute to the maintenance of natural bed patchiness (Townsend and Fonseca in 1998). These are, however, natural processes. Similarly, some dieoffs of seagrass such as the "wasting disease" of the eelgrass (*Z. marina*) in the North Atlantic during the 1930's (Short et al. 1988) and the current demise of *T. testudinum* in Florida Bay have been attributed to a pathogenic form of a marine slime mold, *Labyrinthula zosterae* (Robblee et al. 1991), among other factors. In nature, however, the outbreak of this fungi has not been easy to classify as a cause of seagrass decline as opposed to being a by-product of some other environmentally- or anthropogenically-derived decline in the quality of the seagrass habitat (*sensu* den Hartog 1996).

When human impacts are added to the natural stresses imposed on seagrass beds, disastrous losses of seagrass can occur. Such losses have been documented in Australia (Kirkman 1981, Cambridge and McComb 1984) and southeast Asia (Fortes 1988). In the U.S., large scale losses have been documented in the Chesapeake Bay (Orth and Moore 1981) and in the Gulf of Mexico (Livingston 1987). Significant impacts to seagrass beds in Tampa Bay were documented by Taylor and Saloman (1968), eventually reaching over 50 percent of the historical seagrass cover in Tampa Bay (Haddad 1989). Similarly, 35 percent of the seagrass acreage in Sarasota Bay has been lost as well as 29 percent of that in Charlotte Harbor, Florida, and 76 percent of that in Mississippi Sound (Eleuterius 1987). Pulich and White (1991) reported a loss of 90 percent in Galveston Bay, Texas. Thom and Hallum (1991) report similar ranges of losses from Puget Sound. Large losses of seagrass have also been reported from San Francisco and San Diego Bays (Kitting and Wyllie-Echeverria 1992), the Laguna Madre (brown tide, Onuf 1994), and large-scale damage from propeller scarring has been reported in Florida (Sargent et al. 1995).

Loss of seagrass cover leads to several undesirable and difficult-to-reverse conditions. First, the sediment binding and water motion baffling effects of the plants themselves are lost (Fonseca et al. 1983, Fonseca and Fisher 1986) allowing sediments to be more readily resuspended and moved (e.g., Florida Bay, Thayer et al. 1994). The physical ramifications include increased shoreline erosion and water column turbidity. Seagrass planted in areas with these conditions may not survive due to light limitation from the elevated turbidity. Loss of seagrass, of course, eliminates all important, associated habitat functions (Kikuchi 1980, Peterson 1982).

Much of the documented seagrass loss is due to human-induced reductions in water transparency (Kenworthy and Haunert 1991, Bulthuis 1994; these losses are often not included with other wetland or even seagrass loss statistics). Only in the last few years has it become clear that seagrasses typically require light intensities reaching the leaves of at least 15-25 percent of the light which has penetrated to just beneath the water surface (Dennison and Alberte 1986, Gallegos 1994, Gallegos and Kenworthy 1996). Moreover, the length of time over which a seagrass plant spends at photosynthetically-saturating light intensities too has been shown to be correlated with growth and survival (Dennison and Alberte 1985, 1986, Zimmerman et al. 1991). However, water transparency standards have historically been based on requirements of phytoplankton which may need only ~1 percent of incident light (Kenworthy and Haunert 1991), meaning that there is often no legal mandate for requiring improvement of water transparency to support seagrasses. This absence of technical and legal mandates makes the task of demonstrating the need for restoration of water quality to support seagrasses difficult.

There are many factors that act to reduce water column transparency (*sensu* Dennison 1987, Dennison et al. 1993, Gallegos 1994, Gallegos and Kenworthy 1996). Excess suspended solids and nutrients which enter the water column as the result of poor watershed management combine to reduce transmitted light below that of natural fluctuations, increasing vulnerability to local population extinctions. Suspended solids and water color changes reduce water transparency directly. Nutrient additions, such as from septic systems (Burkholder et al. 1992, Short and Burdick 1996), accelerate growth of light-absorbing algae in the water column as well as benthic macroalgae (den Hartog 1994a,b) and that growing epiphytically on seagrass blades (Sand-Jensen 1977), all of which combine to reduce light availability to seagrasses. Moreover, the seagrass canopy has intrinsic light attenuation effects through mutual shading (Dennison 1987, Enriquez et al. 1992) by the individual plants.

When losses have occurred due to decreased light availability, often only changes in watershed management (such as controlling storm water and sewage discharges) can reverse the trend of decline. Such a reversal in decline is rare but has occurred (Johansson and Lewis 1992). Transplanting into areas experiencing seagrass loss due to decreased water transparency without independent improvements in water quality will only result in the death of the transplants. This is especially problematic in areas where water turbidity may be due to sediment resuspension which arises as a result of seagrass already lost and is therefore not necessarily a current watershed management problem.

Reduction in water transparency is not the only anthropogenic source of seagrass loss (see Phillips 1982 for an early, detailed review). Thermal effluents from electric power plants have caused extensive losses such as those documented at the Turkey Point station in Biscayne Bay, Florida (Zieman and Wood 1975) as well as that associated with the Stock Island (Key West) station (pers. obs.). In the past, dredge-and-fill-associated losses were commonly associated with private sector development but more recently, many losses can be ascribed to public interest projects, such as the replacement of the Florida Keys Bridges (Mangrove Systems Inc. 1985a, Thayer et al. 1985). In addition, the rapidly increasing number of small boats in coastal waters has resulted in the aforementioned widespread damage from propeller scarring (Sargent et al. 1995). Because of the chronic nature of propeller scarring, hull impacts, and, more recently jet ski scour, such damage is likely very difficult to repair by planting (e.g., ferry boat landings in Puget Sound, R. Thom, Battelle Pacific Northwest Lab., Sequim, Wa.), Sargent et al. (1995) recommend a four-point plan to reduce scarring in moderately and severely scarred meadows (defined under their criterion) which includes (1) education of the public as to the nature and scope of scarring impacts, especially in the *Thalassia testudinum* beds which are very slow to recover from impacts, (2) installing channel markers as aids to navigation, (3) enforcing state and federal statutes that address propeller scarring and caused by propulsion systems dredging, and (4) establishment of limited-motoring zones in areas where, due to the extreme shallowness of beds, impacts from propulsion systems would be unavoidable.

A Short History of Seagrass Mitigation and Restoration

Addy's (1947) basic logic was to match planting and harvest site environments, and this remains a fundamental tenet in almost all seagrass planting today. Aside from early interest by Phillips (1960), almost 30 years elapsed before serious attention to planting seagrass developed. It was not until Eleuterius (1975), van Breedveld (1975), Thorhaug (1976), and Churchill et al. (1978) that documents again began to emerge presenting seagrass planting in a guideline format. But even though suitable planting methods have long existed, the track record for successful mitigation of impacts to seagrass beds remains variable (see review by Phillips 1982). Some spectacular failures of seagrass planting (Stein 1984) have created a lasting impression that restoration of seagrass beds is still an experimental management tool. Yet there have also been many successful plantings (e.g., Thayer et al. 1985). Seagrass beds have often been successfully planted and have come to perform much as naturally-propagated beds (Homziak et al. 1982, McLaughlin et al. 1983, Fonseca et al. 1996b). Still it has not been clear what factors are the most important to address to ensure planting success. We had previously thought that seagrass planting was, as Ronald Phillips put it, "a two-edged sword" (R. Phillips, Battelle Labs, Richmond, Wa., pers. comm.), providing a means of ameliorating habitat losses but perhaps encouraging habitat destruction through the mere existence of a possible remedial technique. In our opinion a more conservative trend has emerged. As resource managers and developers have become educated as to the value of seagrass systems and the realities of their costly repair, more emphasis appears to now be placed on impact avoidance and minimization.

Much emphasis was placed on technique development in the late 1970s and early 1980s (see reviews by Phillips 1980, 1982, Lewis 1987, Fonseca et al. 1988, Thom 1990), but relatively little attention was given to developing a management framework within which these techniques could be effectively implemented. As a result, most seagrass mitigation projects failed to achieve the goal of 1:1 habitat replacement (i.e., offset a net loss of seagrass habitat: *sensu* Fonseca et al. 1987c, Fonseca 1989a, but see Merkel 1988a,b), nor have they consistently addressed whether functional equivalency has been achieved (often a permit requirement).

Phillips (1980b) published seagrass planting guidelines that relied on elevation in the tidal zone, current speed, salinity, soil type (sandy, combination, or cohesive) and seagrass species. Decision keys for each coast of the U.S. were compiled. However, with additional research some of Phillips' (1980b) threshold criteria should be changed. He accepted current speeds up to 1.82 m s^{-1} whereas we would strongly caution against planting in current speeds exceeding 0.5 m s^{-1} (see below). Further, Phillips indicated that planting in sandy sediments was a cause for rejection of a planting site, but we have found excellent success in sandy sediments (Lewis 1987, Fonseca et al. 1987a,b,c). Zimmerman et al. (1991) argue that factors increasing root and rhizome anoxia such as cohesive soils recommended by Phillips put seagrass (at least when using bare-root planting methods) under severe physiological stress, a factor to be especially avoided during planting operations. Similarly, Merkel (1992) recommended planting on sandy sediments on the West Coast and avoiding consolidated clays and mudstones (although he [correctly] noted that rhizome extension is slower in coarse sediments). More recently, detailed information on habitat requirements for seagrass (and other submerged aquatic vegetation, SAV) has emerged, but only in well-studied areas. Notable is the work ongoing in the Chesapeake Bay. Batiuk et al. (1992; see also Dennison et al. 1993) provide a detailed synthesis of water quality requirements for SAV (Table 1.2). Based on experimentation and strong correlative evidence of these water quality parameters and SAV distribution, they also developed a series of target water quality conditions that would have to be met to expand SAV distribution by allowing it to colonize greater depths. This study should serve as a model approach to investigate seagrass restoration efforts in other areas. The applicability of these data to other areas is discussed in greater detail under the section entitled "Light Requirements for Transplanting."

Merkel (1992) has developed a field manual for planting eelgrass on the West Coast that includes planning protocols and detailed guidance on planting execution that is otherwise generally lacking in the literature. Aspects of Merkel's report will be reviewed throughout this document. Fonseca (1989a, 1992) published what were essentially Agency checklists for planning and evaluating seagrass plantings; the design of those checklists were the basis for the more comprehensive, yet regionally-specific guidelines published later (Fonseca 1994). The planning, planting, and monitoring sections of this document were adapted from Fonseca 1994: "A Guide to Planting Seagrasses in the Gulf of Mexico." Lockwood (1990) published criteria for placing marinas in eelgrass habitat that extolled impact minimization as the only guideline for mitigation. Based on case reviews of seagrass mitigation projects (Thayer et al. 1985), Thayer et al. (1990) published a preliminary decision matrix that incorporated site selection criteria as well as environmental conditions required for the growth of specific seagrass species.

In general, studies of seagrass restoration and management have only recently become a focus of attention (e.g., Chesapeake Executive Council 1989) and more recently, funding. NOAA's Coastal Ocean Program has focused on these issues for both seagrass and saltmarsh through its Estuarine Habitat Program, C-CAP, and Decision Analyses Series. In conducting our study, we have found the information base for seagrass management difficult to locate. For example, a survey of published literature since 1985 using BIOSIS™ revealed that there were 655 published works on seagrass. This search of the open literature reveals that over the last five years most of the focus in seagrass research has been on aspects of the plant's physiology. This is typical of seagrass research over the last quarter century where interest in plant physiology and seagrass bed-associated fauna have dominated the open literature. Cross-referencing "seagrass" with "restoration" found nine references while "mitigation" provided one reference. From the literature we accumulated directly from journals and solicitation of colleagues, we found that approximately half was found outside the open literature. The literature on the subject of seagrass bed restoration and mitigation is found in the grey literature and is often not subject to the rigors of peer review (but see Batiuk et al. 1992). Another large body of information lies in unpublished project reports, the quality of which are highly variable. We feel that the trend to generate information on seagrass restoration and mitigation for dissemination in forums other than the open literature has been one of the major reasons that seagrass restoration and mitigation is perceived as an experimental tool, when it could be an established management practice.

What are the problems managers face in restoring seagrass beds? Chief among these problems is the tendency to plant seagrass in areas where there is no prior history of their existence (Fredette et al. 1985; unless of course the site was created for the purposes of planting seagrass). The chronic absence of seagrass from a site, especially when there are propagule sources nearby, usually indicates that the site cannot consistently support seagrasses. Ensuring sufficient light, moderate nutrient loads (Batiuk et al. 1992, Dennison et al. 1993, Kenworthy and Fonseca 1996, Short and Burdick 1996) and protecting plantings from disturbance are major considerations for developing a persistent seagrass bed. Planting stock must be chosen so that there are sufficient young shoots and growing meristems to make up for mortality, a ratio that changes dramatically depending on what portion of a seagrass bed is examined, the species, as well as time of year. Most seagrasses are comparatively short-lived and have high natural mortality rates, and suitable growing conditions are needed to allow new shoot generation to compensate for this mortality. Thus, development and incorporation of seagrass demographic information into the management process is a high priority area for research. There are many other caveats that must be imposed to expect successful restoration of seagrass beds. These will be discussed later both in general terms and specifically by region around the country.

Having argued that seagrass mitigation is no longer experimental and should be considered an established management tool, why then place such a priority on conservation? The reason is that while techniques and protocols exist that can produce persistent seagrass beds, the history of the field shows that guidance and protocols are often inconsistently applied. This has resulted in spectacular large-scale planting failures (e.g., the aforementioned Port of Miami expansion project: a multi-million dollar ~200 acre seagrass mitigation which produced only a few acres; Stein 1984). The fact that much information on this subject is conveyed through the grey literature, which does not always circulate widely, has resulted in repetititve mistakes, such as selection of inappropriate planting sites.

MANAGEMENT CONTEXT FOR MITIGATION AND RESTORATION OF SEAGRASS ECOSYSTEMS

Scientists and managers are always faced with uncertainty in decisions regarding ecosystem management. As pointed out by Vitousek (1994) for global environmental change issues, scientists know with certainty that changes are occurring and that they are human-caused. What scientists cannot do is always predict the particular consequences of a given human activity on the environment. However, some trends are obvious and the consequences of inaction can be logically derived. It is irrefutable that extensive loss of seagrass resources have occurred in this country (see previous section), but what are the management options for halting and reversing this decline?

We have compiled this synthesis of seagrass restoration in an attempt to identify reasons for failures and successes which will then allow managers to improve the odds of success in restoring seagrass ecosystems. By acting to mitigate, restore and maintain these resources, managers can offset collateral decline of many ecological functions which we as a society hold important (erosion control, water filtration, fisheries production, and associated aesthetics). However, as the human population grows it is highly likely that losses of these unique plant communities will continue (e.g., Sargent et al. 1995). There are no ecological substitutes for their role in coastal ecosystems.

The critical role that seagrasses play in many coastal environments, coupled with their extensive losses, have created widespread support for their conservation and restoration. The "no-net-loss" policy promulgated by the Executive Branch provided an additional impetus to consider seagrass conservation and restoration. Meanwhile, numerous policy changes have occurred at the state and local levels over the last ten years to support no-net-loss of habitat. Therefore, as an information-based system of judging the value of seagrass ecosystems has emerged over the last decade, the question is no longer whether seagrasses should be protected, but how? When all avenues of protection have failed (e.g., sequencing; the US Army Corps of Engineers-EPA sequence of first seeking impact avoidance and minimization, and then compensatory mitigation, the latter being composed of some combination of enhancement, restoration, creation and under rare circumstances, simply preservation), then active planting may be the only option to avoid a permanent net loss of seagrass.

In order to proceed with discussions of management issues, some terminological clarification is needed. We will utilize the terminologies of Fonseca (1994) which are reprinted in amended form in Appendix A. Particularly, we wish to draw the reader's attention to the differentiation among the terms "restoration" and "mitigation." They are not interchangeable terms. Mitigation refers to activities related to permits (particularly sec. 404 of the Clean Water Act) and embodies a sequence of avoidance, minimization and ultimately, if needed, compensatory mitigation, whereas restoration is simply returning a site to a previous condition. Restoration as used here does not apply to permit-associated planting projects. We will also differentiate the terms "transplanting" and "planting." Transplanting is a subset of planting in that here it refers to harvesting of existing plants whereas planting can involve cultured plants, seeding, or any number of methods. The terms restoration and mitigation set very different constraints on the establishment of performance criteria and the evaluation of compliance (i.e., success). Lewis (1989) defines and differentiates restoration and mitigation as follows:

RESTORATION — "Returned from a disturbed or totally altered condition to a previously existing natural, or altered condition by some action. Restoration refers to the return of a pre-existing condition."

MITIGATION — "...the actual restoration, creation, or enhancement of (functionally equivalent, authors' note) wetlands to compensate for permitted wetland losses."

The term "mitigation" can be used without any modifiers but is often applied to situations more aptly termed "compensatory mitigation." Restoration is a term which generally applies only to planting activities which are not being counted against the destruction of existing habitat. Rather, restoration embraces the concept that anything we can do to right a past loss, a loss for which there may be no litigative recourse to seek damage recovery, is a plus to set against the Nation's balance sheet for no net loss, but not against that of a project with a pending permit to eliminate seagrass. From a management perspective, restoration for the sake of restoration only (properly planned and professionally executed), should be vigorously pursued because it will, if one utilizes the above definition, bring a community back toward previously existing conditions (i.e., it generally cannot make the situation worse).

PITFALLS IN THE MITIGATION AND RESTORATION PROCESS

Compensatory mitigation is a process of questionable merit (Race and Fonseca 1996). Unlike restoration projects which are not necessarily initiated under the 404 permit process, the circumstances under which a compensatory mitigation are initiated have a large potential to make matters worse, because compensatory mitigation usually involves the destruction of existing habitat. The existing habitat is or has been traded for the promise of replacement habitat. With restoration, we are dealing with a past loss for which the responsible party may or may not be identifiable. With compensatory mitigation, the agent of loss and the responsible party are known and sometimes a decision (likely controversial) might be made to trade existing habitat for replacement habitat. Of course when a injury occurs to a seagrass bed outside of the permit process the loss of seagrass habitat occurred without a secure means of mitigating for its loss. However, whether an injury is deliberate or not, if existing habitat is lost, an often tangled negotiation process follows to determine the means by which compensation for that loss will be made. In many instances, the negotiation process can be prolonged, delaying restoration and resulting in larger impacts than might occur if restoration had begun sooner.

There are, however, a number of management decisions that can be made within the permit process to ameliorate a loss in habitat and better approaches the goal of no-net-habitat-loss. Mitigation in its broader definition typically also includes impact avoidance and minimization (the latter term unfortunately implying an acceptable net loss of acreage). In practice, avoidance and minimization are sometimes difficult to achieve. The existence of techniques to transplant seagrass has often been used to justify the destruction of existing, productive habitat (pers. obs.). But as pointed out earlier, this approach has consistently produced a net loss of habitat. This net loss of habitat occurs for a number of reasons, and the permit-associated activities that destroy seagrass beds in the first place typically are long lasting (i.e., creation of channels, bridges, bulkheads). Those activities also often do not allow enough area for on-site planting to offset the loss of habitat. If planting is considered at a location not on the original impact site (off-site restoration or mitigation), that site would preferably not be an area that itself has lost seagrass to some other impact. This is a subtle point that is often overlooked because of the often costly (in time and effort) site history data that must be obtained to make a quantitative evaluation of no-net-loss. The problem works like this: if one permits a loss of seagrass for some form of coastal development (e.g., -1 acre) and plants an equivalent area (+1 acre) onto a site which had previously lost seagrass (e.g., -1 acre) but was not associated with the project at hand, then the net change in habitat is: (-1 + -1) +1 = -1 acre. All that was accomplished was the repair of the original problem at the planting site, but it does not address the loss at the new, most recently impacted site. While there would be no net loss from immediate, present day acreage, the lack of consideration of past losses results in a net loss on a recent historical time scale. The critical question here is at what point in the past do we choose to represent baseline seagrass acreage? Moreover, what if a site chosen for planting does not currently support seagrass? In the absence of site history information, one must then ask why it does not presently support seagrass. This often indicates some inherent difficulty in colonization or persistence of seagrass. The events influencing the colonization process are sometimes difficult to document because they are often aperiodic, acute events (e.g., extreme low tides, storms, migrating rays excavating the bottom). Naturally unvegetated seafloor should not be substituted for vegetated bottom as this typically creates only a transient seagrass bed and alters, not necessarily improves, existing habitat functions. The take-home message is that if one contemplates off-site compensatory mitigation, there are usually few, if any sites available that: (a) can support seagrass growth, and if they do; (b) do not involve habitat substitution; or (c) do not satisfy the no-net-loss goal.This is not to say that previously damaged sites should never be used for mitigation or restoration, they just must be accurately represented in any no-net-loss accounting. As pointed out by Short (Jackson Estuarine Lab., Durham, N.H., pers. com.) in reference to the above description of trade-offs, if no mitigation is done on a previously damaged site, one ends with a -2 acre net loss of habitat instead of -1 acre of loss.

REGIONAL BREAKDOWN OF PERMIT ACTIVITIES DEALING WITH SEAGRASS MITIGATION

Under a Memorandum of Understanding with the U. S. Army Corps of Engineers, the National Marine Fisheries Service, Office of Habitat Protection, comments on development permit requests under Section 404 of the Clean Water Act and Section 10 of the Rivers and Harbors Act. While seagrass restoration has been conducted on an experimental scale along all coasts and within all coastal regions of the U. S., actual mitigation of impacts resulting from Corps of Engineers-permitted activities has been relatively small, and has been greatest in the NMFS Southeast and Southwest Regions. A summary of NMFS-recommended and acted upon mitigation actions by NMFS Region, based on reports received from NMFS Regional Offices as of early 1996, is provided below (note that these regions do not match the ecoregions described later in the text).

NORTHEAST (NE) REGION (Maine through Virginia)

Seagrass mitigation in the Northeast Region of NMFS is in its infancy and, while permits have been reviewed which deal with seagrass habitat, few actions are ongoing. In 1991 NMFS began recommending seagrass mitigation for projects without practicable and feasible alternatives that would damage seagrass habitat. At the time of this report mitigation actions have been considered in New Jersey, Maine, and New Hampshire, but site selection and test planting for a 3-acre mitigation in the Piscataqua River (N.H.) is the only ongoing permit-related mitigation which NMFS has been involved in making recommendations. This has included not only transplanting but also consideration of alteration of bottom topography to achieve appropriate planting depths for eelgrass. Proposals are currently being discussed for a 10-30 acre eelgrass mitigation in the upper Penobscot Bay (Maine).

In addition to supporting the experimental transplanting work that is ongoing in each of the NE states, the NMFS Regional Office has taken a proactive approach to seagrass habitat protection. This has included involvement in the development of seagrass management policies, development of seagrass survey guidelines, encouragement for interagency mapping of seagrasses including involvement of the NOAA Coastal Change Analysis Program (C-CAP) mapping efforts, and the convening of information transfer and education meetings for state and federal agencies on seagrass ecology and transplanting technology.

SOUTHEAST (SE) REGION (North Carolina through Texas including the U. S. Virgin Islands and Puerto Rico)

In the SE Region Habitat Protection Offices reported seagrass mitigation in North Carolina, Florida, and Texas. In each state the permits were obtained or under consideration primarily for channel maintenance or development related to onshore construction. In Texas, however, permit activities relate primarily to petroleum pipeline construction and mitigation of illegal prop-dredging activities. With the exception of Texas there has been little or no monitoring or follow-up to assess the degree of success of the projects. In addition to permit-related activities noted below, field offices of the NMFS have participated in similar management activities noted for the NE as a means of educating state and federal agencies and potential developers of the ecology and sensitivity of seagrass species and habitats.

Between 1985 and 1994 the Habitat Protection Field Office in Beaufort, North Carolina, recommended seagrass mitigation on 5 permits. The direct seagrass damage (i.e., removal of habitat) ranged from 0.23 to 2.0 acres, and the ratio of seagrass planted to that lost ranged from 1:1 to 3:1. However, in one case there was no on-site or in-kind alternative, and oyster reef creation was accepted as an alternative. The seagrasses involved were *Zostera marina* and *Halodule wrightii* primarily, but *Ruppia maritima* also was recommended to be transplanted in one instance. During the 9-year period between 1985 and 1994, a total of 3.25 acres of seagrass habitat were permitted to be destroyed with a requested mitigation of 4.74 acres of seagrass transplantation. Evaluation of the mitigation sites has been carried out in two cases, one demonstrating success and one demonstrating failure.

Florida has the largest extent of seagrasses in the contiguous U.S., followed by North Carolina and Texas (see earlier discussions). Between 1978 and 1994 a total of 167 acres of seagrass habitat have been requested for mitigation by the NMFS Habitat Protection Field Office in Panama City, Florida. These permit requests have generally been the result of new channel construction and port development and have ranged in mitigation acreage from 0.09 to ~200 acres. This latter was the result of a permit for additional development of the Port of Miami. *Thalassia testudinum*, *Halodule wrightii*, and *Halophila engelmanii* have been involved in the recommended mitigation. Based on reports from the Panama City Field Office, the degree of success of these permit-related mitigation has been generally poor and in many cases, unknown.

The Galveston, Texas Field Office of the NMFS Habitat Protection Division reported that there have been 6 major seagrass mitigation activities, almost all in the Laguna Madre, between 1985-1991. A total of 107 acres of seagrass habitat have either been recommended for creation or restoration. These have included filling of unused pipeline channels and associated re-contouring of the bathymetry to downgrading of dredge material islands. In some instances, natural recovery of the site(s) has been recommended while in others transplanting has occurred. The species involved in natural recovery have been *Halodule wrightii*, *Halophila engelmanni*, and *Ruppia maritima*, whereas *Halodule* has been the species of transplant choice. In most instances, oil companies have hired private concerns to monitor the mitigation sites or staff from the Galveston Field Office have had the opportunity to visit the mitigation sites. It appears that site selection and proper bathymetric contouring has occurred because the Field Office reports that with the exception of 22 acres, there has been a mitigation site coverage by seagrasses of between 40-99 percent within a 3 year period by either natural or transplanted methods. Some planted seagrass sites in Florida and Texas are currently being evaluated by National Marine Fisheries Service staff for seagrass and faunal recovery.

NORTHWEST (NW) (Oregon, Washington) AND ALASKA

While research on experimental restoration approaches have been or are being carried out in these two NMFS Regions, both Regional Offices have been involved to only a very limited degree in seagrass mitigation and restoration. For a summary of eelgrass transplanting projects in the Pacific northwest see Thom (1990).

SOUTHWEST (SW) (California, Hawaii, and Pacific Territories)

Similar to other field and Regional Offices in the SE and NE, the Southwest Regional Office has participated in seagrass habitat management at both the permit as well as research and educational levels. They have held state-federal seminars involving the scientific community in discussions on the ecological value, sensitivity, and restoration of seagrasses. In 1991 an eelgrass mitigation policy was drawn up and adopted by NMFS, the U.S. Fish and Wildlife Service and the California Department of Fish and Game that includes recommended transplanting approaches, monitoring approaches, and measures of success that should be considered (see local evidence for seagrass function in Hoffman 1986).

From 1976 through 1993 the SW Region recommended eelgrass mitigation on 25 permits in California while 2 were recommended for *Enhalus acoroides* and *Halodule uninervis* on Rota Island and Saipan Island in the Pacific Territories. With the exception of 6 permits, most mitigation projects have not exceeded 0.1 hectare; the remaining 6 ranged from 0.8-3.8 hectares. Twenty sites have been visited where the mitigation activity had been completed, 11 of which are considered a success by Regional Office staff while 4 have shown a continued decrease in seagrass coverage and the remainder have shown no change in coverage. Overall, the success rate of seagrass planting in this region has been high (Hoffman pers. com.).

COMPARATIVE ANALYSIS OF SEAGRASS PLANTING EFFORTS

At this time we are not aware of any previous analysis of seagrass planting effort in the U.S. that used a comparative method. Therefore, we documented the status of seagrass planting projects from around the country by soliciting information on planting activities from many individuals of whom we were aware had conducted seagrass plantings. In addition, we requested that all National Marine Fisheries Service Regional Offices provide us with listings of all seagrass mitigation projects for which they had reviewed permits under their statutory authority. We also conducted site visits, especially on the West Coast where we were less familiar with planting activities in order to collect additional planting information. Finally we then compiled all references on this subject which we could acquire; this included review of all literature cited by reports and papers we collected.

This is not a complete survey and is complete only through 1995. We undoubtedly have missed some individuals and/or planting projects. Some persons did not respond to our queries. Again the absence of this work from the peer-reviewed literature made it difficult to find the information. The value of this survey then is heuristic, but addresses questions such as "where has effort generally been expended"? What data have been collected? What techniques have been used? How were sites selected and how was compliance and/or performance of plantings determined? How consistently has planting technology been applied?

We also broke down the survey by ecoregions which we have defined for the purpose of isolating practices and caveats peculiar to different parts of the country. Ecoregion is also the basis for the creation of modules where recommendations for planning, planting and monitoring are specifically discussed for each ecoregion. In addition, our original intent was to collect information on coverage rates and shoot addition of individual planting units (PU) from around the country. Any differences in species' coverage and shoot addition rates would aid in the definition of ecological regions for management. However, as our information collection progressed, it became clear that there were insufficient data from most parts of the country to conduct these coverage and shoot rate change analyses. Therefore, we have divided the coastal regions of the country based on our knowledge of growing season. The ecoregions for this report are as follows:

NORTHEAST — Maine through New Jersey: known species present = *Zostera marina* and *Ruppia maritima*.

MID-ATLANTIC — Delaware through North Carolina: known species present = *Halodule wrightii*, *Ruppia maritima* and *Zostera marina*.

GULF OF MEXICO AND THE FLORIDA EAST COAST — Mexico to Cape Sable and north of Jupiter Inlet to Cape Canaveral: known species present = *Halodule wrightii*, *Halophila decipiens*, *Halophila engelmanni*, *Halophila johnsonni*, *Ruppia maritima*, *Syringodium filiforme*, and *Thalassia testudinum*.

SOUTH FLORIDA AND THE CARIBBEAN — South of Jupiter Inlet to Cape Sable and P.R. and USVI: known species present = *Halodule wrightii*, *Halophila decipiens*, *Halophila engelmanni*, *Halophila johnsonni*, *Ruppia maritima*, *Syringodium filiforme*, and *Thalassia testudinum*.

CONTERMINOUS WEST COAST — California to Washington: known species present = *Phyllospadix scouleri*, *Phyllospadix serralatus*, *Phyllospadix torreyi*, *Ruppia maritima*, *Zostera japonica*, *Zostera marina*.

ALASKA — *Zostera marina* and *Phyllospadix* spp.(at least *P. serralatus*).

HAWAII AND PACIFIC TERRITORIES — known species present = *Halophila hawaiiana* (K. Bridges, pers. com.).

We compiled a collection of 138 documents ranging from published, peer-reviewed papers to project reports. Some of these documents were reviews or guidelines of how to transplant seagrass; some were feasibility studies; some were laboratory or mesocosm experiments directed at enhancing transplant technology. Each document was categorized several ways. We first determined where the document originated. Roughly 46 percent of the documents were found in the white literature, 29 percent were unpublished reports, 22 percent in grey literature and ~3 percent were theses (Table 1.3). All together, these papers reported on the fate of over 686,000 planting units of seagrass, totaling ~78 ha of field acreage, that have been monitored.

Over time the publication rate of documents concerning seagrass planting have increased. We found less than 1 percent of the documents published prior the 1960s. In the 1960s we found 2 percent of the documents; in the 70s 21 percent; in the 80s 46 percent; and so far in the 90s, 28 percent of the documents. At this rate the 1990s will produce the greatest amount of documents on the subject of seagrass planting. Some of this increase in publication rate may be that more recently created docu-

Table 1.3. Percent of documents on seagrass planting compiled by literature type.

Literature Type	**Percent of Documents of this Type**
White Literature	46
Report	29
Gray Literature	22
Theses	3

White literature = peer reviewed journal articles.
Report = not peer reviewed.
Gray literature = not in a library circulated journal, may or may not be peer reviewed.
Theses = masters thesis or doctoral dissertation.

ments are easier to locate, but it seems more likely that interest in the subject has grown.

The purpose of the documents varied widely. The largest group was field-research-oriented which comprised ~57 percent of the total. Next were review documents (29 percent), followed by laboratory experiments (~10 percent), and a miscellaneous group (6 percent), which included feasibility assessments, economic analyses, project summaries, and recovery assessments. In addition, there were three planting-associated theses. Most laboratory experiments and review documents were published in the peer-reviewed literature, while only half of the documents presenting new planting data were in the peer-reviewed literature.

Of the ecoregions we constructed, most documents originated from either the West Coast (~26 percent) or the Gulf of Mexico (also ~26 percent) (Table 1.4). South Florida and U.S. Caribbean territories produced ~19 percent, mid-Atlantic region ~18 percent, the northeast U.S. ~9 percent, and Alaska ~2 percent. Studies from other countries (Australia, France, Great Britain, Italy) were also reviewed but not utilized in computation of summary statistics.

The greatest number of planting units have been installed in the South Florida ecoregion (Table 1.5), followed by the northeast, West Coast, and mid-Atlantic states (the latter three regions being almost equal in number of planting units install), the Gulf of Mexico, and lastly, Alaska.

Table 1.4. Percentage of complied documents on seagrass planting presenting field transplanting studies, listed by ecoregion. Values are percent of total. Table does not include studies from outside the U.S., guidelines, reviews, or studies involving freshwater plantings. See section on "Regional Breakdown of Permit Activities Dealing with Seagrass Mitigation," above, for regional boundaries.

Region	**Percent of Documents Found in Region**
West	26
Gulf	25
South Florida	19
Mid-Atlantic	18
Northeast	9
Alaska	2

Table 1.5. Reported area of planted seagrass in square meters and number of planting units deployed in field studies by region and species.

Region	Species	Area M^2	No. PUs
Alaska	*Zostera marina*	?	40
Gulf	*Cymodosa manitorum*[a]	?	150
	Halodule beaudettei[b]	?	150
	Halodule wrightii	8,421	17,956
	Ruppia maritima	1	36
	Syringodium filiforme	591	2,336
	Thalassia testudinum	735	1,087
	Zostera marina	2,025	5,000
Mid-Atlantic	*Halodule wrightii*	2,442	3,924
	Ruppia maritima	56	450
	Zostera marina	63,987	26,960
Northeast	*Zostera marina*	18,449	82,560
South Florida	*Halodule wrightii*	227,639	161,503
	Syringodium filiforme	17,417	20,364
	Thalassia testudinum	332,770	332,239
West	*Phyllospadix torreyi*	?	300
	Zostera marina	102,395	31,262

Alaska = entire coast of Alaska (Ak.)
Gulf Coast = Gulf of Mexico to Cape Sable, Fl. and the Florida East Coast North of Jupiter Inlet to Cape Canaveral (Tex., La., Miss., Ala., Fl.)
Mid- Atlantic = Delmarva Peninsula to North Carolina (Del.Va., Md., N.C.)
Northeast = Maine to New Jersey (Maine, R.I., N.H., Mass., Conn. N.Y., N.J.)
South Florida = South of Jupiter Inlet to Cape Sable, Puerto Rico and the U.S.Virgin Islands (Fl., P.R., U.S.V.I.)
West = Washington to California (Wa., Ore., Calif.).

[a]Probably *Syringodium filiforme.*
[b]Probably *Halodule wrightii*.

PU = planting units.
? = Insufficient data to calculate the area.

The use of different planting methods by ecoregion and seagrass species was also evaluated (Table 1.6). We constructed fourteen categories of planting methods, one of which was an "other" category that contained a number of methods not widely used and includes studies [a category of "unknown"] where the method of planting was not described. Of the fourteen categories, plugs or staples were the most common; ~40 percent of the plantings were done using one of these methods. The next most common was bare root-unanchored sprigs (15 percent), anchors of some sort (8 percent), followed by turfs (7 percent) and peatpots, biodegradable mesh, seedlings and seeds (all at ~5-6 percent each). Unusual or unknown methods accounted for [were employed in] ~2 percent of the plantings. Grids, seed tapes, bagged plants and attachment to boulders, with and without mesh grids, and passive seagrass fragment capture were used in the remaining ~4 percent of the plantings. The Gulf of Mexico ecoregion had the greatest number of planting categories (11), followed by the West Coast (10), south Florida (7), and the mid-Atlantic states (5).

We also compiled the frequency of planting methods used by seagrass species (Table 1.7). *Thalassia testudinum, Zostera marina* and *Phyllospadix* spp. have been transplanted mostly using techniques that involve removal of the native sediment from the root-rhizome matrix (in the case of *Phyllospadix*, there may have been no sediment to remove in the first place). The remaining three species listed in Table 1.7 have been transplanted using sediment-free and sediment-included methods in about equal proportion. Three species, *H. wrightii*, *T. testudinum*, and *Z. marina* accounted for 95 percent of the planting units put in the bottom (26, 21, and 48 percent, respectively). *S. filiforme* composed the remaining 3 percent of the PU while two other species composed ~0.00013 percent of the total number (one paper reported *Halodule beaudetti* and *Cymodocea nodosa* as occurring in the Gulf of Mexico but we suspect these were either *H. wrightii* and/or *Ruppia maritima).* Acreage of planting by species closely followed percentages for PU (Table 1.9). Some seagrass species that have broad distribution have received comparatively little attention to that given *Halodule, Thalassia* and *Zostera*. For example, few studies have been done regarding *Phyllospadix* spp. planting (Phillips et al. 1992), and these involve attachment to large rocks. Aside from Phillips et al. (1992), little else is known regarding *Phyllospadix* spp. planting techniques even though this species ranges along the entire U.S. West Coast (Phillips 1979, Wyllie-Echeverria and Phillips 1994). Turner (1985) provided important data regarding inherent stability and recovery of natural stands that have at least heuristic value for restoration in that the dynamic aspect of the community can be recognized and incorporated into planning (see Chapter 2, Planning). Similarly *Ruppia maritima*, which occurs in every ecoregion, and *Halophila*, of which there may be (based on an incomplete survey) half a million hectares off the West Coast of Florida alone (Iverson and Bittaker 1986), have received virtually no study as to their

Table 1.6. Percentage of all transplanting methods by ecoregion. Values are percent of total. This table does not include studies from outside the U.S., guidelines, reviews, or studies involving freshwater plantings.

Method	**Region**					
	Alaska	**Gulf**	**Mid-Atlantic**	**Northeast**	**South Florida**	**West**
Plug	25	29	20	43	25	12
Peatpot		6	15			8
Turf		19				
Mesh		6	15		3	
Grid		3				
Seedling		3			16	4
Seeds	25		5		13	4
Anchor	25	6			9	16
Sprig	25	11		29	19	20
Seed Tape				14		
Staple		6	45	14	16	24
Boulder						4
MBoulder						4
Other		3				4

Alaska = entire coast of Alaska (Ak.); Gulf Coast = Gulf of Mexico to Cape Sable, Fl. and the Florida East Coast North of Jupiter Inlet to Cape Canaveral (Tex., La., Miss, Ala., Fl.)
Mid-Atlantic = Delmarva Peninsula to North Carolina (Del.,Va., Md., N.C.)
Northeast = Maine to New Jersey (Maine, R.I., N.H., Mass., Conn., N.Y., N.J.
South Florida = South of Jupiter Inlet to Cape Sable, Puerto Rico and the U.S.Virgin Islands (Fl., P.R., U.S.V.I.)
West = Washington to California (Wa., Ore., Calif.).

Planting methods are defined as follows (categories are mutually exclusive):

Plug = tubes as coring devices are used to extract the plants with the sediment and rhizomes intact.
Staple = U-shaped metal staples with attached bare root (no sediment) planting units.
Sprig = bare root planting units (without staples or anchors).
Anchor = any structure used to keep the planting units in the sediment.
Turf = large square sods of seagrass that are ussually extracted with a shovel and planted as is.
Peatpot = a plug of seagrass that is transplanted into a biodegradable compressed peat container.
Biodegradable Mesh = seagrass sewn to a biodegradable mesh fabric and attached to the sediment surface as a planting unit.
Seedling = a newly sprouted seed with one short shoot.
Seed = seeds with no sign of shoots sprouting.
Plastic Mesh Grids = similar to biodegradable mesh except these are plastic (non-biodegradable).
Seed Tape = method of planting seeds using tape that has seeds sticking to it; the tape is then rolled out along the sediment surface.
Boulder = *Phyllospadix torreyi* is attached to boulders.
MBoulder = *P. torreyi* is attached to mesh and then attached to boulders.
Other = rarely used methods and includes studies where the method was not stated in the document.

Table 1.7. Percentage of transplanting methods by seagrass species. Dashed line separates methods that transport associated sediments (above line) from those that do not (below line).

Method	Species					
	Hw	Pt	Rm	Sf	Tt	Zm
Plug	32			42	11	21
Peatpot	9		25	8		6
Turf	7		25		3	2
Mesh	9		25		4	4
Grid	3					
Seedling					21	2
Seed					14	6
Anchor	6			17	7	11
Sprig	3			17	25	19
Seed Tape						2
Staple	23		25	17	7	23
Boulder		?				
MBoulder		?				
Unknown	3					2

Hw = *Halodule wrightii*
Pt = *Phyllospadix torreyi*
Rm = *Ruppia maritima*
Am = *Zostera marina*
Tt = *Thalassia testudinum*.
Sf = *Syringodium filiforme.*

Planting Methods are defined as follows (categories are mutually exclusive):

Plug = tubes as coring devices are used to extract the plants with the sediment and rhizomes intact.
Staple = U-shaped metal staples with attached bare root (no sediment) planting units.
Sprig = bare root planting units (without staples or anchors).
Anchor = any structure used to keep the planting units in the sediment.
Turf = large square sods of seagrass that are usually extracted with a shovel and planted as is.
Peatpot = a plug of seagrass that is transplanted into a biodegradable compressed peat container.
Biodegradable mesh = seagrass sewn to a biodegradable mesh fabric and attached to the sediment surface as a planting unit.
Seedling = a newly sprouted seed with one short shoot.
Seed = seeds with no sign of shoots sprouting.
Unknown = the method was not stated in the document.
Plastic mesh grid = similar to biodegradable mesh except these are plastic.
Seed Tape = method of planting seeds using tape that has seeds sticking to it; the tape is then rolled out along the sediment surface.
Boulders = *P. torreyi* is attached to boulders.
MBoulders =*P. torreyi* is attached to mesh and then attached to boulders.
? = insufficient data to calculate a percentage.

Table 1.8. List of experimental parameters and the percentage (in descending order) that were incorporated as a data collection or as independent variables in field transplant studies. Pre-survey = the site selected as a transplant site was surveyed prior to transplanting for its suitability to sustain a transplant.

Experimental Parameters	Percent of Documents Using this Parameter
Pre-survey of site	62
Planting method	45
Post-survey of site	27
Depth	26
Cost analysis	22
Fertilization type	21
Season	21
Faunal study	18
Planting unit spacing	17
Tidal zone	15
Energy regime	14
Donor survey	12
Sediment particle size	9
Enclosure	8
Shoot numbers	8
In vitro propagation	8
Genetics	6
Light intensity	5
Bioturbation	3
Burial recovery	3
Apicals	1
Salinity	1

Planting method = different methods of transplanting were tested for their effectiveness.
Post-survey of site = the effect of transplanting on the site location was evaluated.
Depth = effects of different depths on transplanting success was determined.
Cost-analysis = the total cost of the transplanting was determined.
Fertilization type = effects of fertilizers on transplanting was evaluated.
Season =effects of time of year on transplanting was evaluated.
Faunal study = fauna was sampled in transplanted beds.
Planting unit spacing = the effects of different spacing of planting units was evaluated.
Tidal zone = effects of different tidal zones on transplanting was examined.
Energy regime = effects of energy regime on transplanting.
Donor survey = there was a study conducted on the recovery of the transplant donor bed.
Sediment particle size = effects of different sediment size on transplanting.
Enclosure = effects of enclosure devices on transplanting.
Shoot numbers = effects of different planting unit shoot numbers on transplanting success.
In vitro propagation = growing seagrass in the laboratory to be transplanted.
Genetics = genetic experiments on transplanted seagrass were conducted.
Light intensity = effects of various light levels on transplanting success.
Bioturbation = bioturbation effects on transplanted seagrass.
Burial recovery = effects of sediment burial on transplanted seagrass.
Apicals = effects of the presence, absence, or different numbers of apicals in planting units.
Salinity = effects of different salinity on transplanting.

Table 1.9. List of ten most common parameters recorded in monitoring of transplant studies. Some studies considered more than one parameter.

Monitoring Parameter	Percent of Studies with this Parameter
Irregular frequency monitoring	74
Percent survival (PU)	65
Shoot counts	55
Shoot density	53
Percent cover	47
Leaf length	29
Leaf width	12
Rhizome length	6
Directm mapping	3
Biomass	3

Irregular Frequency Monitoring = irregular time intervals were chosen for follow-up monitoring of a transplant site.
Percent Survival = percent of planting units (PUs) that survived were monitored.
Shoot Counts = direct counts of planting unit shoots was conducted.
Shoot Density = density of the planting units was monitored.
Percent Cover = time zero area was known and considered 100 percent cover so that future areal coverage could be compared as a percent of that original coverage.
Leaf Length = leaf lengths were measured directly.
Leaf Width = leaf widths were measured directly.
Rhizome Length = total length of living rhizome.
Direct Mapping = actual mapping of the planting units for the area covered.
Biomass = weight of a given area of seagrass.

ecological role in the coastal zone. Although there has been little in the way of focused attention on development of planting techniques for these latter two species, we expect that existing methods such as plugs or peatpots may have promise (see Chapter 2, Planting).

There are also some incidental plantings of which we are aware. We know that *Halophila decipiens* was transplanted in 15 meters of water on St. Croix, U.S.V.I. in 1986 (authors unpubl. data) using mini-staples constructed of 130-pound test wire leader; plantings spread and apparently persisted to the end of the normal growing season. Harrison (1990) has also transplanted *Z. marina* in British Columbia using unattached shoots, cores, and by attaching shoots to re-bar (*sensu* Kelly et al. 1971). *Phyllospadix* was planted in the Monterey Bay Aquarium, California. Indoor small tank exhibits had poor survival but plantings lodged under rocks and experiencing mild simulated wave conditions in the aviary persisted for several years (Monterey Bay staff, pers. com.). Similarly, *Thalassia* has been grown in the coral reef exhibit at the Smithsonian Institution in Washington, D.C. for several years but with great logistic cost.

Not only did purposes of these papers vary widely, so did the design parameters of the studies. Table 1.8 describes the various parameters that were manipulated for documents reporting new field planting data (66 documents total: reviews and lab experiments excluded). Twenty-four different parameters were examined. Preliminary surveys of some environmental conditions at the planting site was the most common design feature: ~62 percent of the papers performed some pre-planting evaluation of the site. Slightly less than half of the papers tested planting methods. Only ~25 percent of the papers continued to survey some environmental conditions after plantings were installed, making it very difficult to establish any linkage between plant performance and episodic events. Planting depth (a rough surrogate for light availability, but also potentially related to frequency of emersion) was at least noted, if not a factor tested for influence on plantings in approximately 30 percent of the papers. Tidal zone (as opposed to some sea level-normalized depth measure) was also noted in 17 percent of the papers, but these data were not as specific as depth data. Together, however, water depth and tidal zone considerations were in 47 percent of the papers. Cost analyses, comparisons among planting season, and fertilizer effects were aspects of project design in ~20 percent of the papers. Comparative faunal assessments, effects of PU spacing, physical energy on the site, and recovery of plants at the donor site were parts of project designs in 12-18 percent of all studies. An additional 12 parameters were examined in the papers we reviewed but were never included in more than 10 percent of the papers.

What is interesting here is not so much what was either manipulated or noted but the proportions of what was not; that is, data that were considered relevant varied tremendously among studies. Thirty-eight percent of the papers did not consider or at least did not report what information was used to choose a planting site. Of those reporting, 33 percent simply used the criteria of no vegetation present which when used alone has been previously described as an unacceptable criteria (Fredette et al. 1985, Fonseca et al. 1987c, Fonseca 1989a, 1992, 1994) because selecting unvegetated areas with no known history of seagrass cover disregards the fact that any one of several mechanisms may be at work maintaining that level of patchiness (e.g., waves, currents, bioturbation). There is a rich body of literature on the role of habitat heterogeneity on ecosystem function that would have to be ignored to recommend converting naturally unvegetated areas to vegetated. Thus, in addition to being a high-risk planting area, planting in such an environment temporarily substitutes one habitat type for another. Therefore, based on our survey, 24 percent used what we consider to be an appropriate site selection criteria (the site having been previously vegetated but was now barren, although there are caveats to this criteria; see Planning chapter). Approximately 19 percent of the plantings were on dredged material while 20 percent were on unvegetated spaces adjacent to existing seagrass.

Few other factors were consistently integrated into design plans. Only 17 percent surveyed fauna after planting. Twelve percent considered the impact of harvesting on donor seagrass beds (i.e., monitored recovery of donor site). Less than 10 percent of the studies manipulated time 0 shoot number in a planting unit (generally as an attempt to determine optimal planting unit size).

Interestingly, two parameters besides percent PU survival that we have long recommended as being critical baseline monitoring data (Fonseca et al. 1982, Fonseca 1989a,1992,1994), number of shoots PU^{-1} and percent cover of the bottom, were only in ~53 and ~47 percent of the papers, respectively. We have recommended these forms of data collection because, when combined they describe many aspects of planting viability. In contrast, shoot density, a parameter over which there is little control, was used as a performance criteria in ~51 percent of the studies. Recent findings (Fonseca et al. 1996b) also suggest that macroepibenthic faunal abundance in planted seagrass beds asymptotes at comparatively (to natural beds) low shoot densities (as little as one third of natural beds), indicating that it might not be relevant to require shoot density in a planted bed to equal that of natural beds to support faunal densities equivalent to natural (but see performance criteria suggested by Short (1993 p. 51). Although some lower-than-ambient threshold shoot density may be suitable for generating faunal equivalence, lower shoot densities may not provide a sufficient buffer to population fluctuations of the seagrasses themselves. Thus, the issue of demographic status of the seagrasses of restored vs. natural beds is only beginning to be evaluated.

Most disturbing was that less than 7 percent of the papers actually provided quantitative data on two of the most critical limiting parameters known for seagrass planting success, light regime and bioturbation. Although depth and tide zone were frequently recognized as important factors, the absence of direct measurements of light means that depth and tide zone data are not easily extrapolated because we do not know the transparency of the water column. We can look up information on tidal amplitude and periodicity, but the interaction of light and tides on seagrass growth is only now being modeled (Zimmerman et al. 1994, Dennison and Kirkman 1996, Koch and Beer 1996), although these papers suggest the interaction of tidal amplitude and light availability to accurately predict site suitability based on transmissivity data.

Apart from those parameters that were monitored and/or manipulated, a total of ten parameters were actually utilized as measures of planting performance and/or success (Table 1.9). The percentage of PUs surviving was the most common criteria but was reported in only ~66 percent of the papers. However, ~74 percent of the papers varied the frequency of monitoring after planting over the course of their respective investigations (e.g., contrast fixed interval monitoring with a study that conducts monthly sampling for the first year then shifts to biannual monitoring for following years). The duration of monitoring in the papers we reviewed ranged from zero to eight years with a mean and median of ~1.5 years. During these monitoring periods the frequency of monitoring was also highly variable, again ranging from zero to an equivalent of 30 times y^{-1}. The average frequency of monitoring was 4.6 times y^{-1}.

Fonseca (1989, 1994) has recommended that early, frequent (usually quarterly) monitoring be performed for the first year after planting followed by less frequent (e.g., biannual) monitoring. Despite problems with changing temporal scales in analysis (see section "Scale and its Role in Defining Seagrass Habitat"), we continue this recommendation because many, but not all (particularly plantings with high initial loss of PUs) of our successful experimental plantings followed a sigmoidal population growth curve; initially high, exponential growth with low mortality followed by a balancing of natality and mortality of shoots which leads to an asymptote of plant density. Past recommendations for this monitoring strategy (Fonseca 1989a, Fonseca 1992, Fonseca 1994) actually agree well with at least the mean monitoring time values of the papers reviewed. Similar frequencies of monitoring were recommended by Merkel (1992), of time 0, 3, 6, 12, 24, and 36 months but with an additional recommended survey at 60 months. Choice of 3 years for monitoring resulted largely from compromise in that permit monitoring is rare (Race and Fonseca 1996) and shorter monitoring periods increase the possibility of acquiring monitoring compliance. So, for a given planting, how long should monitoring proceed in order to judge planting performance? Taken together with the average monitoring period of 4.3 y, and the fact that only 10 percent of the papers we surveyed achieved an ideal 100 percent cover, indicates that previous suggestions of 3-year monitoring by Fonseca (1989a, 1992, 1994) may be a serious underestimate of the time required to document project success; times in excess of 5 years may be more appropriate.

What is probably the most documented parameter in natural beds, seagrass biomass was only measured in ~3 percent of the papers, perhaps because it is a destructive sampling technique. Also, several measures of the plant's morphology were used frequently to determine planting performance (Table 1.9). We view these criteria suspiciously; seagrasses are often phenotypically plastic, and variation in plant shape and size is only loosely linked to functional attributes of seagrass beds at this time (Bell et al. 1991, Fonseca et al. 1996a) although morphology has been linked to significant genetic differences (Fain et al. 1992). We find it disturbing that simple parameters such as survival and coverage were not more universally recorded. From the low replication of criteria among studies, it is no wonder that quantitative performance and compliance thresholds, when they appear in mitigation plans, vary so tremendously (Thayer et al. 1985). Moreover, some papers used irreproducible units such as "scoopfuls" and "bucketfuls" to describe sampling units. Such vague planning criteria should not be used by resource managers.

The results of monitoring efforts have revealed some unexpected trends regarding success. To analyze this, we chose two categories, the final reported percent PU survival and the percent of the target area covered that was reported at the conclusion of a paper. Of 53 papers that reported percent PU survival, the median percent PU survival was 35 percent; mean 42 percent; standard deviation = 29.9; coefficient of variation = 70 with a distribution heavily skewed to lower percent survival (Sk = 0.35), suggesting adoption [use] of the median value (Figure 1.5). Roughly 5 percent of the plantings reported 100 percent PU survival. We found 27 papers that reported percent of the target area covered. The median percent area covered was 40 percent and was closer to the mean percent area covered of 42 percent; a standard deviation = 31.2; but still with a high coefficient of variation = 75 and a distribution again skewed to lower coverage amounts (Sk = 0.41) (Figure 1.6). We should point out that some of the variance in the data also results from areas such as southern California enjoying generally very high success rates (approaching 100 percent). The reasons for that success rate may have to do with quiescent settings for planting, high experience level and perhaps, comparatively low bioturbation levels. However, on a national scale, only approximately 10 percent of the plantings achieved 100 percent cover within the monitoring period. Thus, these data indicate that replanting is a consistent requirement of seagrass operations unless substantial initial overplanting is conducted to compensate for anticipated losses. Moreover, low initial survival rates may explain why seagrass plantings often produce less acreage than originally planned, suggesting that initial PU survival levels should be held to high standards to help ensure achieving target acreage.

An extreme interpretation of these findings would be that based on the median survival (a planting should have an overplanting ratio of approximately 3.0). In other words, if you wished to ensure that 100 planting units will survive, 300 should be

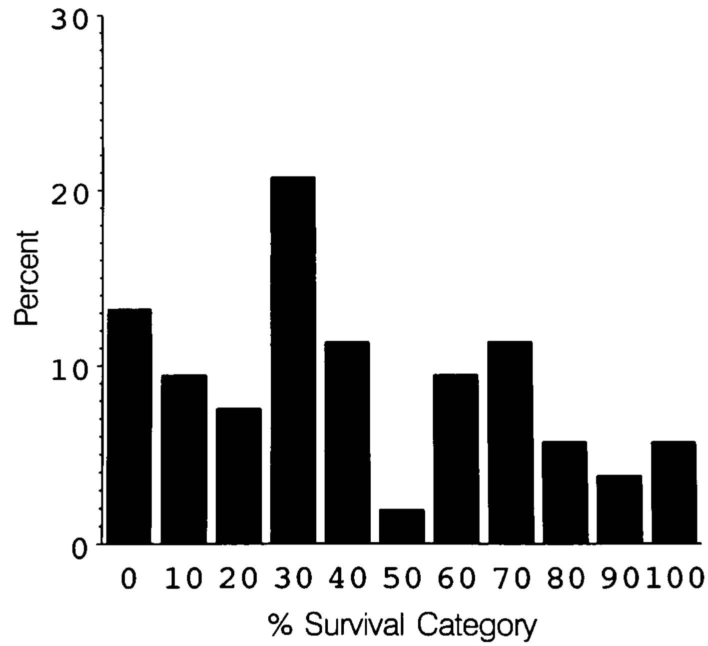

Figure 1.5. Frequency distribution of percent planting unit survival from the documents surveyed nationally. Y-axis = percentage of the survival values falling in the percent survival categories on the X-axis (10% increments).

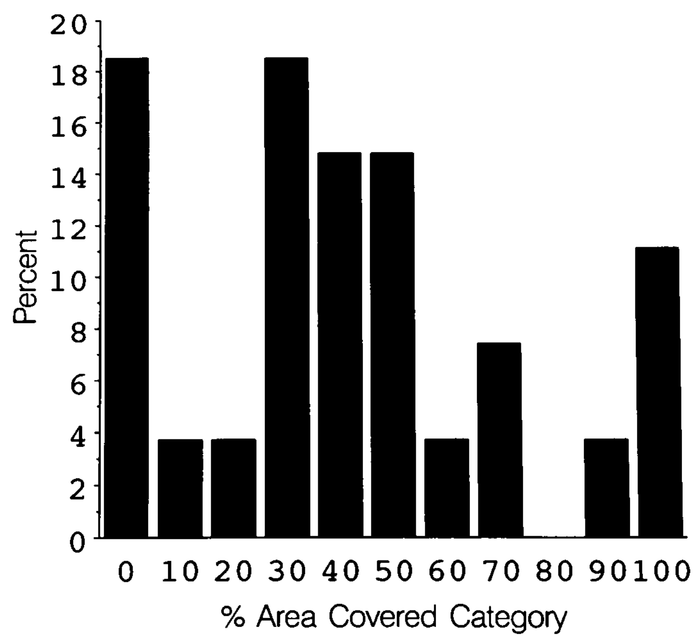

Figure 1.6. Frequency distribution of percent area covered by plantings from the documents surveyed nationally. Y-axis = percentage of the area covered values falling in the percent area cover categories on the X-axis (10% increments).

planted. Similarly, based on this national average, to ensure the required area of seagrass bed to be generated, a replacement ratio of ~2.5 units of area planted to 1 unit of area lost is needed to meet a no-net-loss criteria (i.e., 1:1 replacement ratio). We conclude that this is an extreme interpretation because many plantings used to compile these statistics were conducted on sites that would have been expected to produce patchy seagrass beds in any event. Also, many sites were chosen that violated recommended site selection criteria which would skew the distribution toward low survival and coverage. If site selection criteria are employed as described later, it is possible that these replacement ratios could be made much lower.

Managers must be cognizant of the different sources of planting failures and judge planting proposals under strict criteria. The practice of seagrass bed mitigation should not be questioned based on a failure in judgment on the part of someone who performed a planting. Such human failures must be separated from failures of the approach as a whole in order to responsibly assess seagrass planting as a mitigative tool (Fonseca et al. 1994). The key is to determine what made some plantings so successful and others so marginal.

Monitoring as recommended in the past (e.g., Fonseca 1989a, 1992, 1994) does not lend itself to determination of agents of planting loss. Only sophisticated monitoring equipment with high frequency recording capacity could hope to detect environmentally-induced losses. Acute and capricious events such as bioturbation and vandalism are even more difficult to determine with complete certainty (although use of exclosure cages may go far in suggesting the influence of bioturbation, Merkel 1988a, Fonseca et al. 1994). Therefore, the agents of loss among these studies cannot accurately be presented as a ranked set. However, based on our observations in the field, one might speculate that most failures occur from improper site selection (see criteria for site selection, below) and execution. From our experience and conversations with others (not to mention some published findings: Mote Marine Lab. & Mangrove Systems Inc. 1989; Merkel 1988a,b; Fonseca et al. 1994), we conclude that once a site has been appropriately selected under the criteria described below (e.g., previous history of seagrass cover, etc.) the primary agents of loss vary between bioturbation, acute storm events, algal smothering, and vandalism.

These compilations indicate that most of the planting experience is centered in the southern and western parts of the U.S. Also only a few species are regularly utilized in mitigation projects. Given the widespread impacts to seagrass ecosystems, concern that the absence of these other species from the literature indicates that impacts to those species goes unnoticed. Either that or these plant communities may not be receiving sufficient protection under current management practices. These survey data also indicate that there has been extensive experimentation with planting methodologies, and it appears that only a few are consistently employed and, again, only for a few species. The concentration of planting effort in Florida and the West Coast may be due to comparatively high development pressures in these areas. Although high habitat loss rates also occur in the mid-Atlantic states ecoregion, the proximity of research laboratories that have historically focused on seagrass to those estuaries may also explain the concentration of work in that ecoregion.

ARE PLANTED SEAGRASS BEDS FUNCTIONALLY EQUIVALENT TO NATURALLY-OCCURRING BEDS?

What is "functional equivalency"? In a general sense, this means that a restored or mitigated system attains functions the same as those of an unimpacted system in a similar setting. Seagrass beds have many functions (*sensu* Wood et al. 1969), some of which may be more difficult to restore than others. As is the case with much of biology, the answer to the question of functional equivalency is both "yes" and "no." We tend to take the stance that if an area has recovered equal or greater acreage than that which was lost, and that area persists with the same seagrass species, a planted seagrass bed can become equivalent, but not identical to a natural, unimpacted bed. Our stance is not universally accepted. Equivalent means "equal to" but is sometimes taken to mean "identical." However, since no two samples of any natural ecosystem are ever truly identical, some subjectivity comes into play, both in terms of the degree of equivalence and the appropriate functions to measure. The problem then is what drives the subjectivity? A developer may interpret functional equivalency of their mitigation project in far more general terms than a trained biologist. What then are the relevant parameters by which to document equivalency?

According to our comparative analysis of the literature, thirty-three different parameters were used to describe success. This indicates the broad definition of functional equivalent — practitioners obviously target many different factors and differ in their opinions when ranking importance of these factors. Moreover, there is conflicting guidance from the literature regarding the rate at which planted beds take on attributes of natural, undisturbed beds. Brown-Peterson (1993) and Montagna (1993) conclude that attributes of planted seagrass beds were still not equivalent to natural ones after 31 and 14-17 years, respectively. Similarly, Smith et al. (1988a) found that planted beds did not provide equivalent bay scallop habitat over a growing season. Hoffman (1988) concluded that one-year old *Z. marina* plantings in San Diego did not support some fauna at levels exactly equal to that of natural beds, although some of differences were small. In contrast, Nessmith (1980), Homziak et al. (1982), Fonseca et al. (1990, 1996 a,b), and Wyllie-Echeverria et al. (1994b) found that faunal abundance and composition in planted beds approached that of natural beds within 2-3 years.

Much of this discrepancy among studies may be the result of intrinsic differences among natural reference sites and planted areas, the organisms chosen to evaluate recovery, and different worker's interpretation of what constitutes a difference. Because of the tremendous variability among natural beds, we question the efficacy of precise numerical comparisons in an interpretation of planting success; comparisons that include estimates of variance might be more appropriate. For example, distance and/or isolation of planted sites from natural beds will cause some differences (Bell et al. 1988, 1992). Brown-Peterson (1993) compared fish communities among planted and reference sites but the sites were located on opposite sides of a barrier island lagoon. Montagna (1993) compared beds established both by natural recolonization and planting in scraped-down dredged material islands with relatively open areas. Thus, there is some question as to whether differences among planted and natural treatments were the result of planting or of innate differences due to the physical setting. However, Montagna (1993) points out that most studies suggesting faunal equivalency have focused on more vagile macrofauna such as fish whereas certain infauna (e.g., clams) may not colonize as quickly. Kenworthy et al. (1980) and Homziak et al. (1982) found rapid colonization of a planted *Z. marina* site by scallops and meiofauna as did Wyllie-Echeverria et al. (1994b) for salmon prey (largely meiofauna). McLaughlin et al. (1983) concluded that recolonization by a wide variety of macrofauna occurred in planted *Thalassia* beds within only a few years. Similarly, Fonseca et al. (1990) found that after experiencing widespread failure of a planted area, the same site then naturally colonized by seed and supported a macrofaunal community not statistically different from adjacent planted sites within six-months of the onset of seed germination.

More recently, Fonseca et al. (1996 a,b) found that *H. wrightii* and *S. filiforme* beds planted on 0.5 m centers in Tampa Bay developed fish, shrimp and crab density and composition statistically indistinguishable from nearby natural sites within three years. One interesting aspect of that work was the relation of animal density to plant density (Figure 1.7). The seagrass density at which animal density in planted beds equaled ($p < 0.05$) that of natural beds was only approximately one-third of the mean natural bed shoot density. That density can be obtained within one year. They found that although linear models could account for approximately 65 percent of the variance of animal density as a function of plant density over time, a non-linear, asymptotic relationship between natural-log transformed animal density and seagrass areal shoot density was apparent (Figure 1.7). Although transformation of a straight line

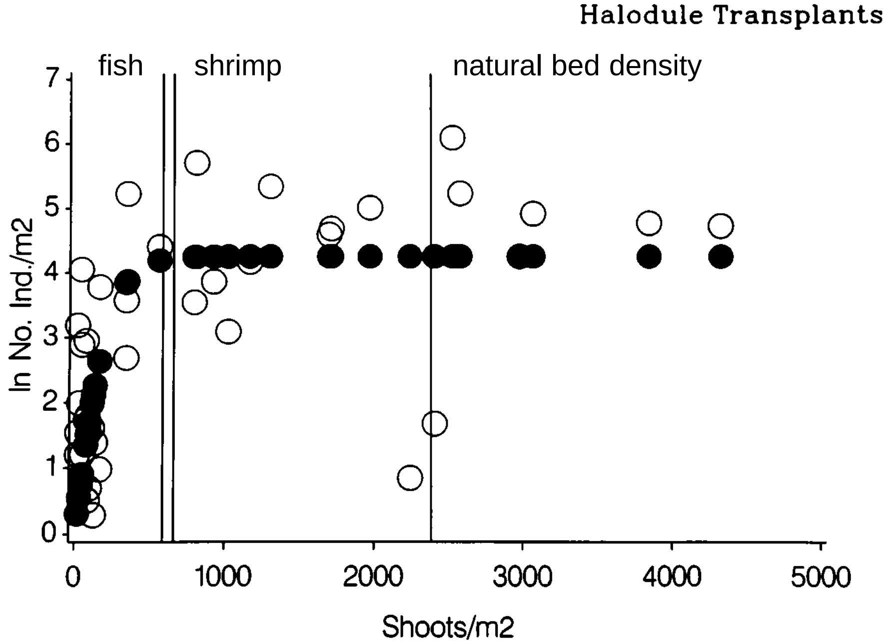

Figure 1.7. Natural-log transformed faunal density plotted against areal shoot density. Open circles=observed points. Closed circles=predicted points using an asymptotic function. Vertical line (VL) "shrimp" or "fish"=areal shoot density at which densities of these animals first became not significantly different among planted and natural versions of that seagrass species; "natural bed density"=the study-wide average areal shoot density of natural beds of that seagrass species. Taken from Fonseca et al. (1996b).

will yield some asymptotic tendencies, they are not enough to account for the pattern observed. The fact that animal density had an asymptotic relationship with shoot density implies that monitoring shoot density over time may be an inexpensive diagnostic parameter for determining a threshold planting success in terms of fauna. Short (1993) found similarly rapid colonization of *Z. marina* plantings in New Hampshire by a wide variety of fauna. Therefore, monitoring shoot density over time would be much less costly than direct measures of faunal communities. In order to justify use of only plant data in assessing some aspects of planting success, however, the temporal relationship among shoot density and faunal community structure must be collected from planted beds across a broad geographic range.

Although some ecological attributes may return quickly after planting seagrass, there is still a measurable period of time until the system has attained full function. The loss of ecosystem production in the time between when a seagrass bed is damaged and functions are restored has long been an issue of concern to managers. This loss of production has been termed "interim loss." The manner in which this loss has been calculated varies widely, as will be discussed later under sections on planting. We raise the point here because in many instances a reckoning of functional equivalency among planted and reference sites is made in an attempt to recoup interim loss of various living resources.

Seagrass species substitution is another issue that has bearing on the question of functional equivalency among planted and reference beds. Although much of the temperate U.S. is dominated by one seagrass species, *Z. marina*, subtropical areas and more recently the Pacific Northwest must contend with the functional ramifications of substituting one seagrass species for another (Pawlak 1994). There are few data to guide this decision. Temporary species substitution has been suggested for subtropical species (Fonseca et al. 1987c) where faster-spreading species such as *H. wrightii* and *S. filiforme* are planted as predecessors to recover areas previously dominated by the much slower-spreading *T. testudinum*. In that circumstance the reasoning was that interim loss of ecosystem functions could be minimized by establishing any form of seagrass coverage. Based on their work in Tampa Bay, Fonseca et al. (1996b) have suggested that differences in macroepibenthic faunal communities among these three subtropical species may not be as great as previously inferred (Stoner 1983). One reason for the differences among studies could be that Fonseca et al. (1996b) focused on unit area of seagrass bed-based surveys while others had compared animal populations among seagrass species weighted by attributes of habitat complexity, such as leaf surface area. While acknowledging that such faunal assessments should not be construed as indicators of all ecosystem functions (e.g., nutrient cycling, bed stability), the findings of Fonseca et al. (1996b) support the notion that seagrass species substitution to ameliorate interim losses of some ecological attributes may be a legitimate means to an end where that end is eventual replacement of the seagrass species that was damaged.

There is another tack to take in assessing whether planted seagrass beds provide resource functions equivalent to those they are intended to replace. On a much simpler level, we are not aware of any study that suggests that a seagrass bed is *not* a highly productive habitat. Therefore, we may infer that if one can produce a desired amount of seagrass habitat which persists over time that many ecosystem functions eventually will be restored. Whether a planted bed is the exact replacement for another seems to us to be an inappropriate question. It will depend upon what one considers to be an important ecosystem function and how much of it must be replaced to be considered "equivalent." Utilizing generally accepted significance levels (i.e., $p < 0.05$) to detect differences among two unique portions of the ecosystem is the most objective and scientifically acceptable means of testing differences. Of course, such differences are certainly affected by decisions of sampling gear, its size, the bias of that gear towards certain fauna and size classes, and temporal and spatial density of sampling. For example, the differences reported by Brown-Peterson (1993) and Montagna (1993) may well be within the range of year-to-year and site-to-site variation if longer periods of time were sampled and across wider areas (the choice of sampling scale being a powerful determinate of our perception) (e.g., crab densities: Fonseca et al. 1996b). Given the spatial and temporal variance in animal numbers, we feel it very difficult to justify additional planting based on relatively small differences in faunal attributes as compared to unplanted areas; using less stringent significance levels may be acceptable as well (i.e., $p < 0.10$).

In general, we believe that planting is a success if the acreage is planted, persists, and eventually (if not immediately) leads to replacement of the same resource functions of the seagrass species that were damaged. By success we imply functionally equivalent to natural beds. Again, we stress that the use of acreage and persistence as diagnostic features of planting success needs additional geographic replication.

As mentioned earlier, this view of assessment of planting success is, of course, not universally held. Short (1993) established other criteria for mitigation success for a project in New Hampshire. There, based on the recovery observed in his planted beds, he suggested that for eelgrass plantings to be considered initially successful, they should cover 30 percent of the planted area in one year. In addition, he stipulated that 40 percent of the following parameters be attained by plantings within one year: seagrass primary production, shoot density, leaf area, percent cover, and continuity (i.e., meaning that 40 percent of the plantings have coalesced). Moreover he stipulated that fish and infaunal assemblages constitute 25 percent of the following ecological parameters: presence of dominant species and total numerical abundance as compared to nearby natural beds. The values put forth by Short (1993) appear to coincide with findings for Tampa Bay (Fonseca et al. 1996b) in that faunal recovery will be closely linked with planting success and persistence.

Another reason for using plants rather than fauna as a metric of planting success is that one can envision scenarios where faunal recruitment to a bed could be inhibited by recent natural events such as storms or local pollution sources that may or may not equally affect a reference site. Conversely, we know of no evidence where an otherwise unimpacted natural seagrass bed has not supported high faunal density and diversity. Thus, if the faunal/seagrass relationship is not necessarily reciprocal, use of fauna alone may not be as easily tracked as the plants themselves, which may be suitable to infer the development of collateral faunal resource functions. At the very least, following plants alone is arguably less expensive than faunal data collection and may represent the most realistic data collection effort except in heavily subsidized restoration efforts. A pragmatic reason for accepting simple measures of acreage and persistence is that few plantings are well-monitored and enforcement of non-compliance with permit conditions is sporadic (*sensu* Race and Fonseca 1996). The parallel between seagrass presence and fauna we feel is strong enough to accept some metric of seagrass alone as a viable indicator of functional recovery. Although additional research must be conducted to strengthen the seagrass-faunal function link on a broader geographic basis, our research suggests that monitoring the seagrass itself is a very useful option for assessing restoration/mitigation success.

Combinations of parameters have also been suggested as an appropriate metric for gauging success of seagrass plantings (F. Short, Jackson Estuarine Lab., Durham, NH., pers com.). Combining factors that are known to affect faunal abundance, such as shoot size, density and water depth (taken together as a measure of habitat complexity), may be a way to provide a better means of indirectly comparing functions among planting, impact, and reference sites, especially when they occur in different geographic and/or physical settings. Short (Jackson Estuarine Lab., Durham, N.H. pers. com.) has proposed canopy volume:

(shoot density * canopy height; [m/m^2= #shoots/m^2 * m/shoot])

because it is a value that should change more slowly than shoot density alone and thus be more tightly coupled to a greater range of ecosystem attributes beyond faunal development (e.g., current speed reduction, change in sediment composition, nutrient cycling). Moreover, the canopy volume metric can be obtained with non-destructive methods and, unlike shoot density, would likely not exhibit overcompensation responses with some seagrass species (e.g., *Syringodium filiforme*: Williams 1990, Fonseca et al. 1994) which do not accurately reflect long-term recovery from injury to a seagrass bed.

Another factor in assessing functional equivalency is habitat size. We do not know if there is any relationship between the size or shape of a seagrass bed and its functional attributes; this is true for both planted beds and natural beds. This is an area of study needing much additional work. From our experience, however, even very small patches 1-2 m^2 of seagrass in the Beaufort, N.C. area has significantly greater numbers of fish, shrimp, and crabs than found in adjacent sand areas (unpubl. data). Moreover, Murphey and Fonseca (1995) found that on a unit area seagrass basis, even beds in the range of 30-40 percent cover had penaeid shrimp densities virtually indistinguishable from that of continuous cover, 100 percent beds. These data taken together with years of personal observation of seagrass beds (both planted and natural) have left us with the impression that even these very small, isolated patches provide resource functions comprising much of that observed in more extensive, unbroken coverage beds on a unit area basis. Thus, even very small patches of seagrass deserve protection.